THE BAPTISTE LAKE STUDY

TECHNICAL REPORT

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EXECUTIVE SUMMARY

The primary purposes of the Baptiste Lake Study, carried out between March, 1976 and March, 1979 were to determine the trophic status of the lake, and to develop and employ methods for assessing the impact of existing or future shoreline and watershed development on that status. In order to determine the trophic status of the lake an intensive limnological study was undertaken; the latter included weekly observations of phytoplankton carbon fixation, phytoplankton standing crop, taxonomic changes and changes in the physico-chemical environment. In order to assess existing cultural impacts to the lake, the sedimentary record was analyzed to determine any historical changes in the fertility of the system. In order to predict the effect of future development, a detailed nutrient budget was constructed for the lake and a comparative nutrient loading analysis undertaken.

Baptiste Lake drains a large watershed and, as such, has a large external loading of nutrients. Prior to European settlement this loading, primarily from a boreal mixed-wood forest, was large enough to ensure that the lake was in a eutrophic condition. This interpretation has been confirmed by the sediment core analysis.

The clearing and conversion of forested land to agricultural use has had a measurable impact on stream nutrient loadings, the net result of which has been to elevate the trophic status of the lake to a more eutrophic condition. By comparison, the estimated nutrient loadings from cottages were relatively small,

although the question of nutrient bio-availability was not assessed in this study.

The average total phosphorus (TP) export coefficient for forested lands was 0.14 kg ha^{-1} yr^{-1}, and that from agricultural land was 0.27 kg ha^{-1} yr^{-1}. These data are similar to the mean coefficients presented by the US EPA (1974) and Uttormark (1974).

The average total nitrogen (TN) export coefficient for forested land was 2.50 kg ha^{-1} yr^{-1} and for agricultural land 2.13 kg ha^{-1} yr^{-1}; these differences between land use types were not statistically significant. These coefficients are lower than those measured in many other locations, and the TN pools differ significantly in composition. The TN loads were primarily composed of organic nitrogen (approximately 75% for agricultural streams and 90% for forested streams).

The TN:TP ratios in both stream basin types were high and suggest phosphorus deficiency, however, inorganic N:P ratios were below 7.2 and suggest nitrogen deficiency. This conundrum will remain until some understanding of the bio-availability of the organic nitrogen component is attained.

The atmospheric input of nutrients was measured in devices similar to those described by Likens et al. (1979) for the Hubbard Brook Study. The annual input of TP was estimated at 30.4 mg m^{-2} yr^{-1}, which is lower than that occurring in many other regions. Most of the deposition occured during spring, summer, and autumn, with little input over winter.

Groundwater flows into the lake were estimated by a mathematical model developed by Crowe and Schwartz (1981). They estimated that approximately 13% of the annual hydraulic loadings could be derived from groundwater. However, because of the diversity of hydrostratigraphic units surrounding the lake basin, each with different hydrochemical characteristics, we did not attempt to estimate nutrient loading from this source. Direct evidence for groundwater seepage into the lake was produced as a result of discrete chemical testing in the north basin.

Baptiste Lake has one seasonally stratified basin, and one temporarily stratified basin. The lake is partially meromictic as a result of incomplete spring mixing, and displays many of the characteristics of the classical eutrophic system. Hypolimnetic oxygen depletion occurred in the deep, southern basin and this resulted in a large accumulation of nutrients in the hypolimnion, presumably from sediment release. TP concentrations exceeded 500 mg m^{-3} near the bottom in late summer and this hypolimnetic pool probably enriched surface waters as the thermocline was depressed. Heavy blue-green algal blooms occurred during late summer, with peak surface chlorophyll a concentrations observed in the high hundreds of mg m^{-3}.

The shallow, northern basin stratified for approximately three months each summer; this was also accompanied by oxygen depletion and nutrient accumulation. Isothermal conditions developed by late August in this basin so that the hypolimnetic nutrient supply enhanced the summer blue-green bloom, as opposed to enhancing the autumn pulse of phytoplankton as occurred in the southern basin.

The seasonal pattern of phytoplankton succession is similar to that described for eutrophic systems by Reynolds (1982). The open water season lasts approximately six months, extending from late April to early November. The spring diatom assemblage is dominated by *Stephanodiscus astraea*, with the pyrrhophyte *Peridinium sp*. making brief but important contributions to the biomass during May. An unidentified unicellular blue-green, now believed to be broken colonies of *Microcystis aeruginosa*, was also significant. An early summer minimum during June was characterized by a mixed community which included *S. astraea*, several cyanophytes, and the cryptophyte *Cryptomonas ovata*. The cyanophytes dominated from early July to mid-September, however, there was a shift from domination by *Anabaena circinalis* and *Microcystis aeruginosa* to *Gomphosphaeria naegelianum* and *Aphanizomenon flos-aquae* in August. The large pyrrhophyte *Ceratium hirundinella* was also present during July and August and was extremely important in terms of total cell volumes. During the month of October, the phytoplankton was again dominated by the diatoms *Melosira granulata* and *Stephanodiscus astraea*, but also included the cryptophyte *Cryptomonas ovata*.

The annual carbon fixation rate was determined by the radiocarbon method and estimated at approximately 302.5 g C m^{-2} yr^{-1}. Most photosynthetic activity was concentrated in the open water season, although some under-ice activity was recorded in late March as the photoperiod increased.

Extreme spatial heterogeneity, or patchiness, was observed in surface chlorophyll *a* distributions in Baptiste Lake. The degree of patchiness increased as the concentration increased, and these data pointed out the importance of statistical sampling design in the accurate trophic assessment of productive lakes.

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1.0 INTRODUCTION

The Baptiste Lake Study was initiated in the winter of 1976 by the Water Quality Control Branch of the Pollution Control Division, Alberta Environment, with two primary objectives in mind:

1) to determine the trophic status of Baptiste Lake and to assess the impact that shoreline/watershed development has had on that status, and
2) to test and/or develop methods for quantitatively predicting the impact that shoreline development could have on various other lake systems in the province. This would complement the concerns of other provincial agencies involved in planning for lake and shoreland use, who have been restricted by the lack of suitable techniques for use in assessing lake "development capacities".

The trophic status of a lake can be defined arbitrarily by such criteria as its nutrient loadings, nutrient concentrations, phytoplankton species composition and standing crop, annual phytoplankton primary production rates, and dissolved oxygen stratification characteristics. In order to define these and other parameters for Baptiste Lake, a detailed limnological monitoring program was established and conducted between March 1976, and March 1979. In addition, two relatively undeveloped lake basins in the same physiographic region were examined in order to place the data from Baptiste Lake in a more complete perspective. These were Ghost Lake and the northern basin of Island Lake.

Alterations in the trophic status of Baptiste Lake as a result of European settlement and shoreline development were elucidated using two different procedures:

a) the nutrient budget approach, which defines the inputs and outputs of selected nutrients to the lake, and assesses the significance of different sources, and

b) the paleolimnological approach, in which the minerology and structure of the sediments, their organic and inorganic chemical constituents, and the morphological remains of the organisms preserved in the sediments are analyzed to permit interpretations about the past conditions of the lake.

Methods for quantitatively predicting the impact of shoreline and watershed development on lake ecosystems are based upon the phosphorus modelling sequence and have only been developed in the last decade. One of the most significant contributions in this area has been that of the OECD Eutrophication Project (1981), which has examined the relationships between phosphorus and nitrogen loads to lakes and the algal biomass that results. This project and other studies have shown that phosphorus is a key element controlling the excessive fertility of many fresh waters.

The effect of various development scenarios on Baptiste Lake was dealt with in a previous report (Trew et al. 1978) and will not be discussed extensively here. This report is a detailed compilation of all in-lake observations and nutrient loading measurements, and includes a discussion of the application of the phosphorus modelling sequence to the management of Baptiste Lake and others lakes in the prairie-parkland and boreal mixed-wood forest biomes.

2.0 PHYSICAL SETTING

2.1 LOCATION

Baptiste Lake (54° 45'N; 113° 33'W) is located approximately 16 kilometers west of the town of Athabasca (Figure 1) and is situated within the interior plains physiographic region of the province. The Baptiste Lake watershed (Figure 2) drains an area of approximately 30,891 hectares through 12 tributary streams and 13 diffuse runoff areas, with one outlet stream (Baptiste Creek) flowing from the lake into the Athabasca River. A small tributary intercepted the outlet in a diffuse zone close to the lake shoreline and consequently was monitored to determine its influence on the observed hydrological and chemical regime of the outlet. Land elevations range from 578 m above sea level at the surface of Baptiste Lake to an elevation of more that 690 m west of the lake (Figure 3). The principal morphometric parameters for Baptiste Lake are presented in Table 1.

2.2 GEOLOGY

The geology and geological history of the Baptiste Lake watershed have been studied in detail by Crowe and Schwartz (1981). Their results are based upon an analysis of data collected during field programs (test-holes drilling, augering, air photos) and a review of existing reports. The following sections summarize the work conducted by the same authors.

The uppermost bedrock unit underlying Baptiste Lake is the Upper Cretaceous La Biche Formation consisting of dark green-gray

Table 1. Morphometry of Baptiste Lake.

Length	7,296 m
Maximum Width	2,324 m
Mean Depth	9.3 m
Maximum Depth	27.5 m
Lake Volume	85,526 dam^3
(at 578.35 m Geodetic Survey or Canada)	
Shoreline Length	26,441 m
Watershed Area	30,891 ha
Lake Surface Area	917 ha

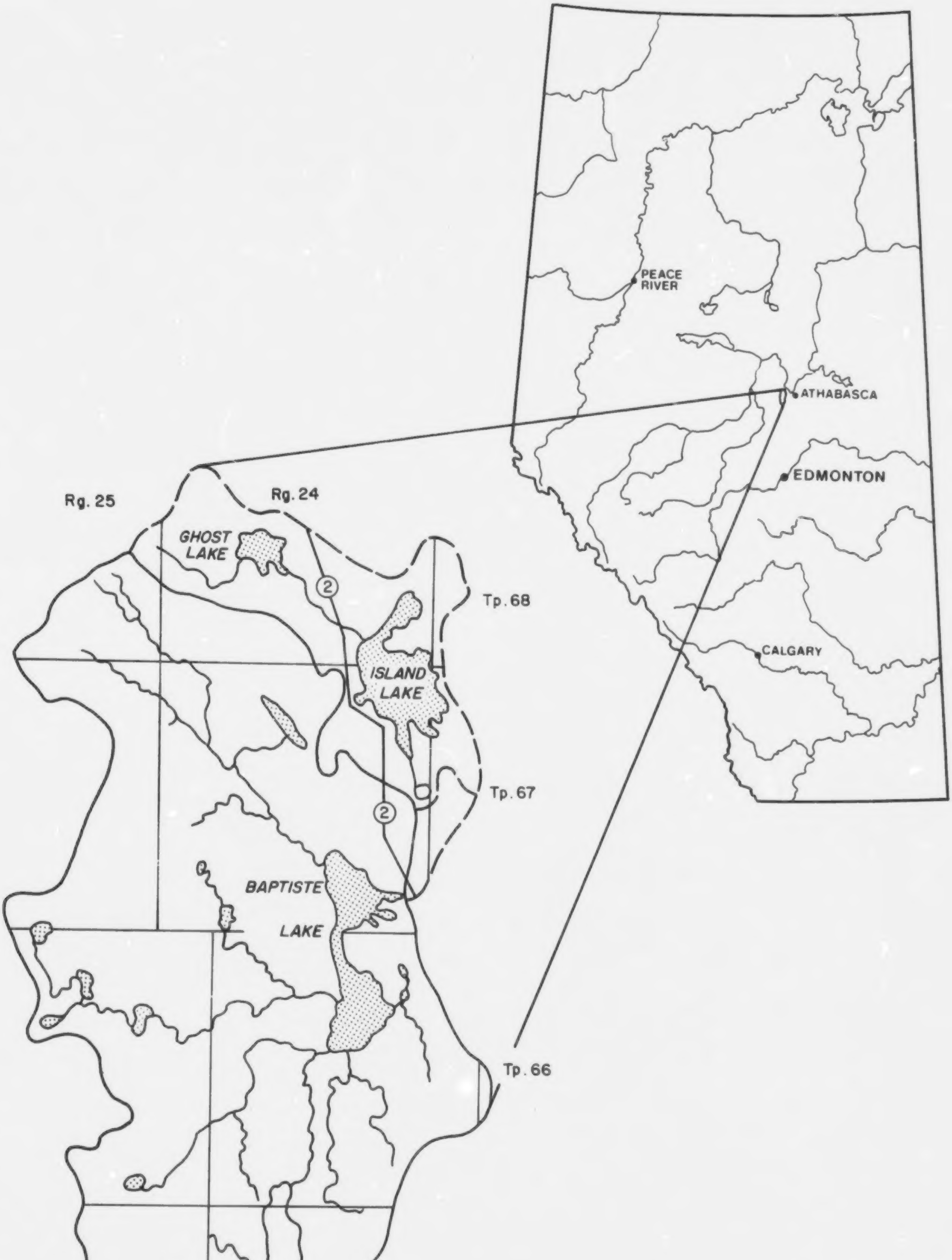
PEACE RIVER
ATHABASCA
EDMONTON
CALGARY
Rg. 25
Rg. 24
GHOST LAKE
ISLAND LAKE
BAPTISTE LAKE
2
2
Tp. 68
Tp. 67
Tp. 66
Tp. 65

FIG. 1 BAPTISTE LAKE STUDY AREA LOCATION PLAN

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Water Quality Control Branch

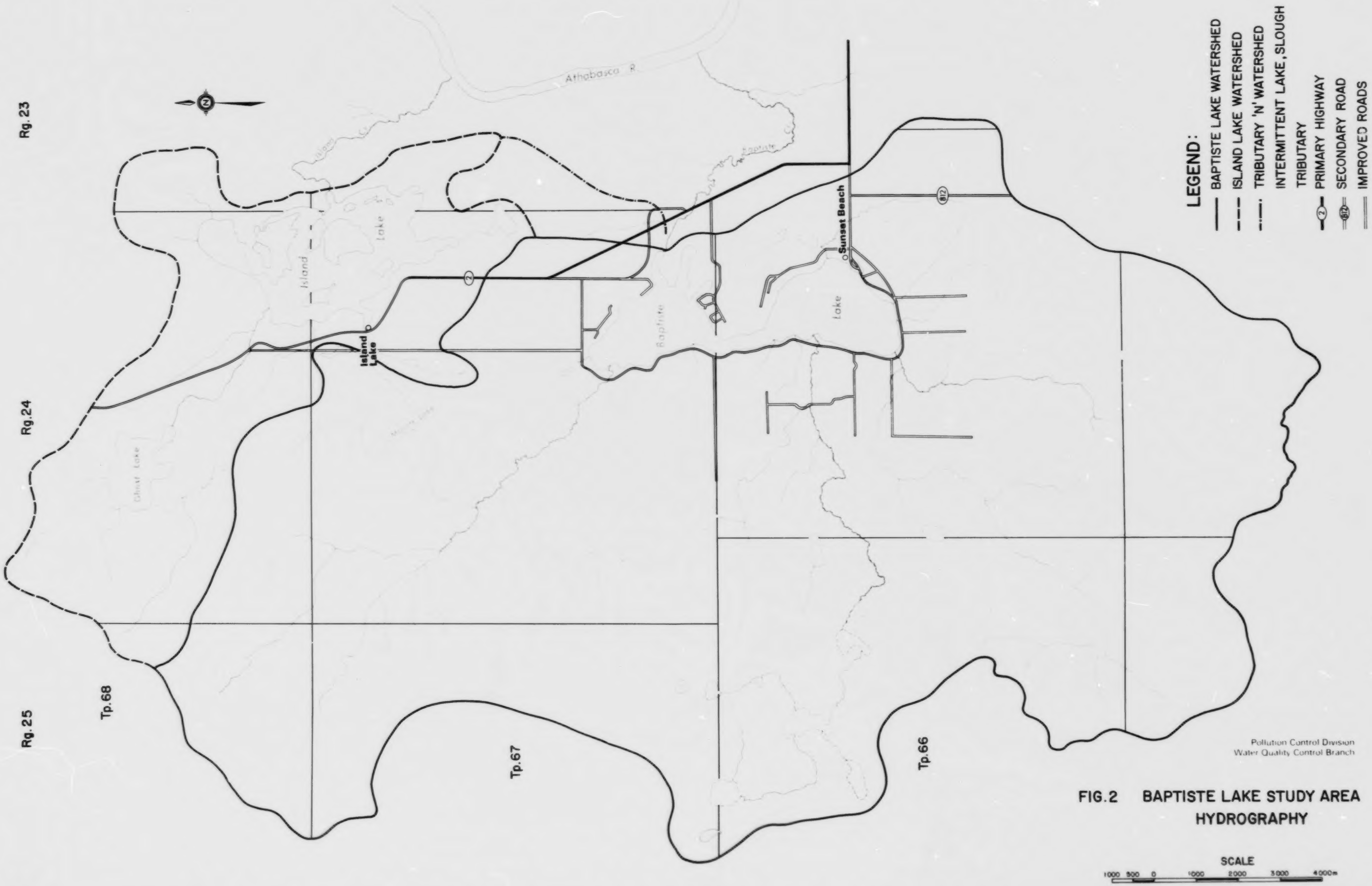
LEGEND:
BAPTISTE LAKE WATERSHED
ISLAND LAKE WATERSHED
TRIBUTARY 'N' WATERSHED
INTERMITTENT LAKE, SLOUGH
TRIBUTARY
PRIMARY HIGHWAY
SECONDARY ROAD
IMPROVED ROADS
Rg. 23
Rg. 24
Rg. 25
Tp. 68
Tp. 67
Tp. 66
Athabasca R.
Baptiste
Island Lake
Island Lake
Baptiste Lake
Sunset Beach
Ghost Lake
2
812
N
SCALE
1000 500 0 1000 2000 3000 4000m

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FIG. 2 BAPTISTE LAKE STUDY AREA HYDROGRAPHY

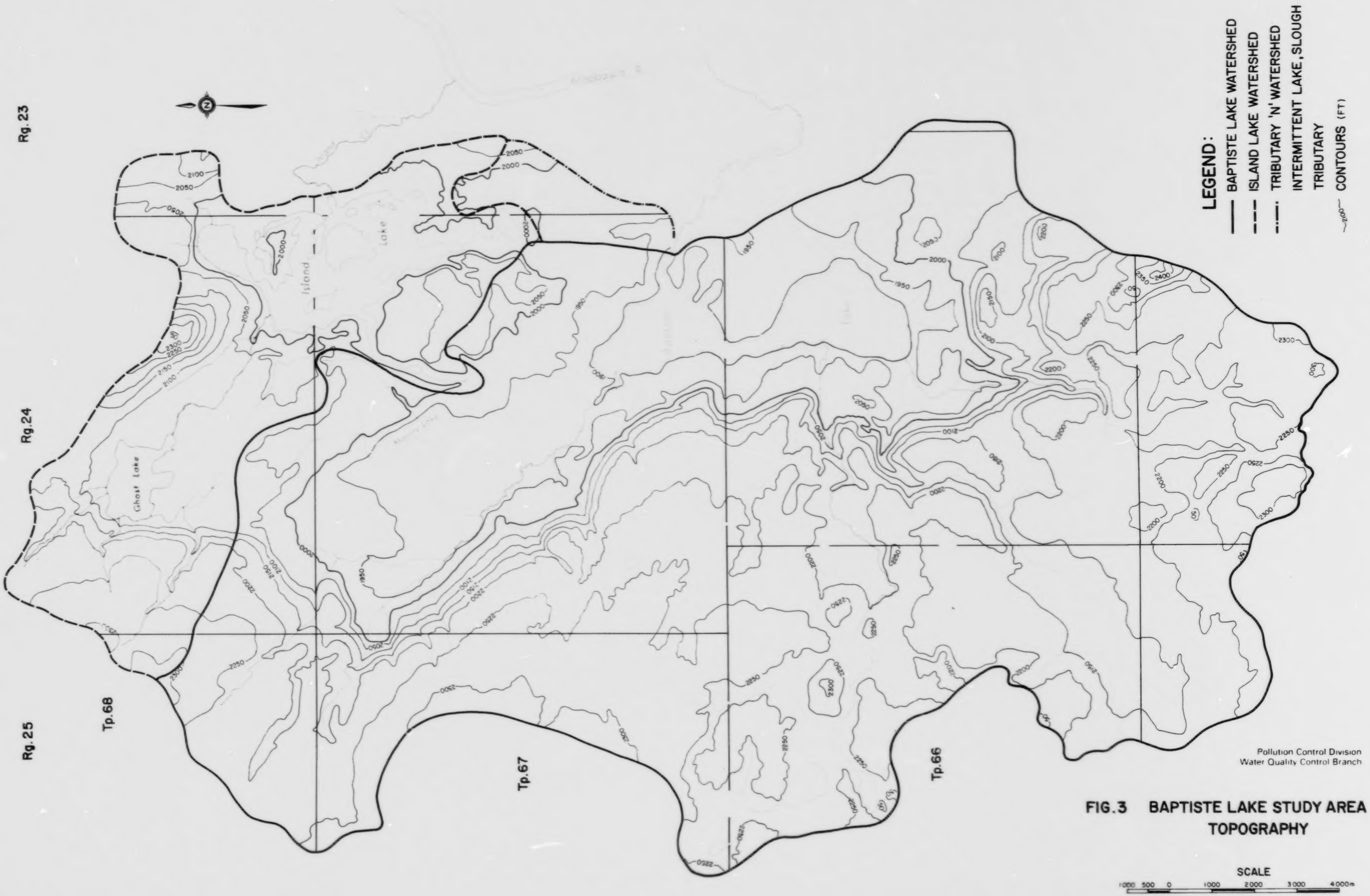
Rg.23
Rg.24
Rg.25
Tp.68
Tp.67
Tp.66
N
LEGEND:
BAPTISTE LAKE WATERSHED
ISLAND LAKE WATERSHED
TRIBUTARY 'N' WATERSHED
INTERMITTENT LAKE, SLOUGH
TRIBUTARY
2100 CONTOURS (FT)
Island Lake
Ghost Lake
1950
2000
2050
2100
2150
2200
2250
2300
2350
2400
SCALE
1000 500 0 1000 2000 3000 4000m

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FIG.3 BAPTISTE LAKE STUDY AREA TOPOGRAPHY

marine shales. The hills to the west of Baptiste Lake are formed by the exposure of a younger unit, the Upper Cretaceous Wapiti Formation consisting of argillaceous sandstone, bentonitic mudstone, and coal. The lake is situated in a bedrock depression which is part of a pre-glacial buried bedrock valley (Figure 4). This valley trends from southwest to east at an elevation of approximately 525 m above sea level (15 m below the lake bottom). A tributary buried valley trending northwest to southeast converges with the main channel beneath Baptiste Lake. Bedrock elevations in the surrounding area typically exceed 575 m west of the lake and 560 m to the east.

Five genetically distinct moraine units were identified. The upland area to the west of the lake is covered by a stagnant ice moraine with typically hummocky and "doughnut" topography. The broad, low relief area north of Baptiste Lake represents an area where glacial thrusting occurred. It is bounded by scarps on its west and northeast margins. Most of the material that originally existed within this depression was transported down-ice virtually intact and was deposited as ice-thrusted moraine in the form of hills south of Baptiste Lake. The material which remained northwest of the lake is composed of moraine and alluvium which was reworked and considerably mixed by the ice thrusting. Thrust-fluted moraine, found north and west of the lake, has been reworked and remoulded by an ice advance into poorly defined fluted and drumlinoid forms. Low relief, undifferentiated ground moraine is found southeast of the lake. Evidence of ice thrusting processes is common on the Canadian prairies and may be the principal mode of formation of many prairie lake basins (Crowe and Schwartz 1981(a)).

Glacial till is the dominant surficial deposit and ranges in thickness between 30 - 100 m. The tills are clay-rich and contain very few pebbles, although some sandy and stoney layers are present. The source of the till has been identified by Kjearsgaard (1972) as the underlying La Biche Formation. Overlying the till in some areas are lacustrine deposits, which are found as thin layers (0.2 - 3.0 m) of dark grey, stone-fee silt. These deposits are formed as a result of sedimentation in a pro-glacial lake basin, and are good indicators of the past size of Baptiste Lake. The lacustrine deposits in the Baptiste Lake watershed are found along the outlet stream and in the area at the southeast portion of the lake. The alluvial and aeolian deposits, which represent a small percentage of the surficial deposits, are medium to coarse grained sands. The alluvial deposits, which are formed by deposition from running water, are primarily found at the north end of the lake. The aeolian sands, which are deposited by the wind, are found at the southwest end of the lake.

Soils in the watershed are rated as Class 4 and 5 dark grey wooded soils, and have severe to very severe limitations that restrict the range of crops (Canada Department of Forestry 1965).

2.3 LAND USE

Morphometric parameters for stream basins and diffuse runoff areas in the Baptiste Lake watershed were measured by planimetry from standard 1:50,000 topographic maps. The relative proportion of land use types was also determined by planimetry from maps prepared by the Regional Planning Section, Alberta Municipal Affairs.

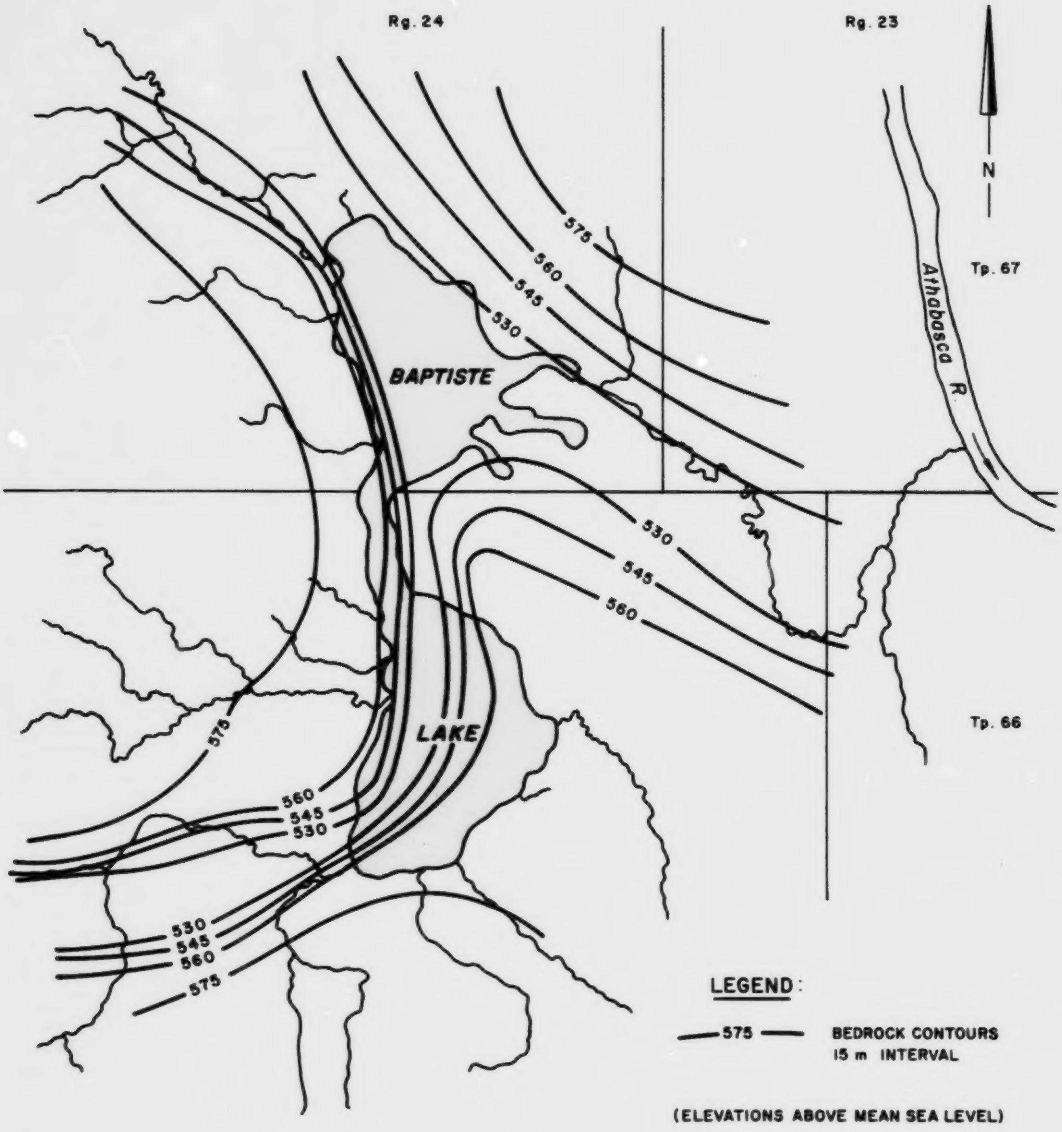

Rg. 24
Rg. 23
N
Tp. 67
Tp. 66
Athabasca R.
BAPTISTE
LAKE
575
560
545
530
LEGEND:
575
BEDROCK CONTOURS
15 m INTERVAL
(ELEVATIONS ABOVE MEAN SEA LEVEL)

FIG. 4 BAPTISTE LAKE STUDY AREA
BEDROCK TOPOGRAPHY

Pollution Control Division
Water Quality Control Branch

The watershed of Baptiste Lake was first surveyed between 1904 - 1907 by Hugh McGrandle, D.L.S., at which time only a few Metis families were living in the area. They were reportedly growing "some hay and potatoes" for their own use on very small parcels of cleared land (i.e. 5 acres), otherwise the watershed was essentially undisturbed (Dominion Land Surveyor's Notebooks, Alberta Transportation, Edmonton). Homestead lands were made available subsequent to the survey and much of the present agricultural land was apparently first broken in the period up to 1915 (Stone 1970).

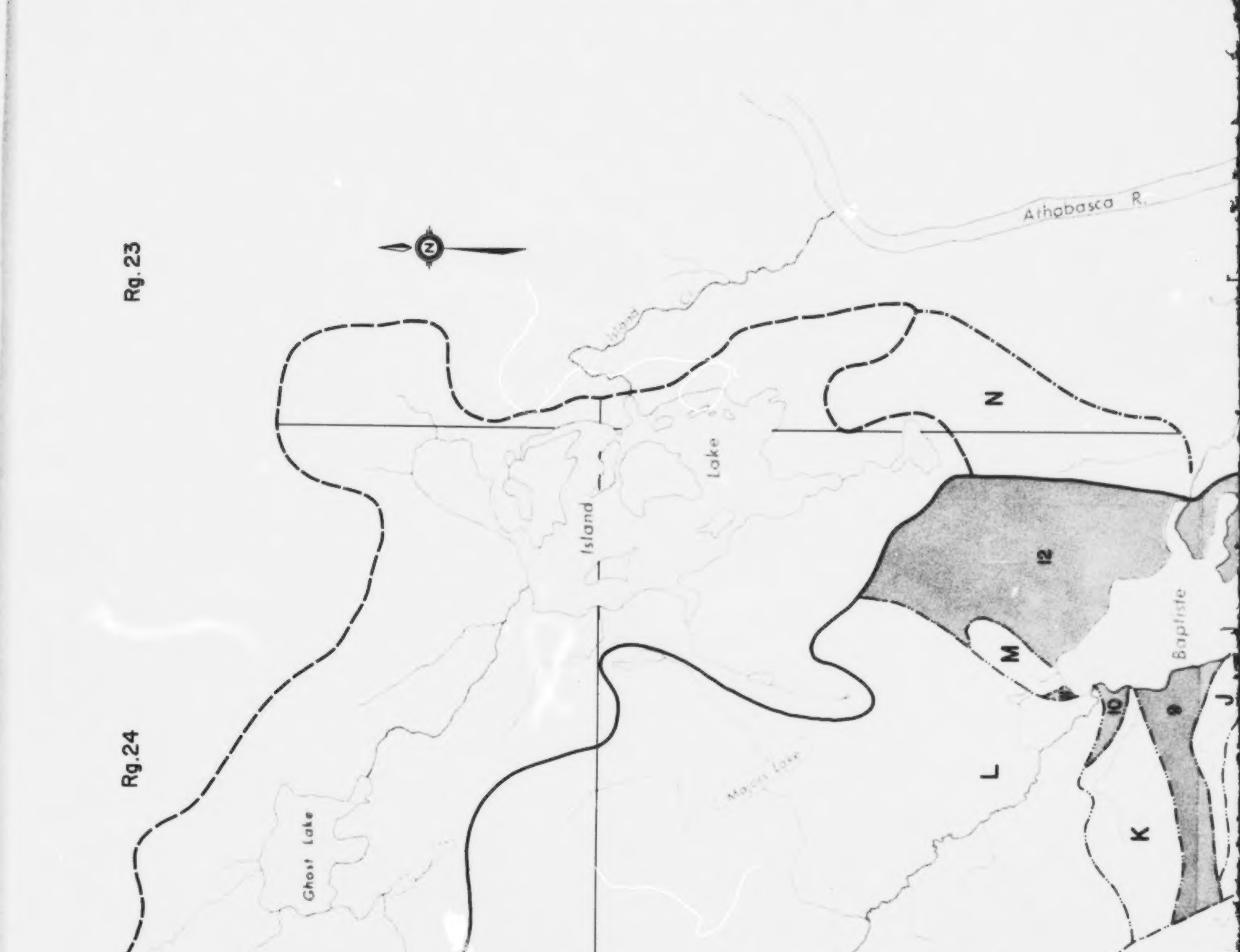
The 12 stream basins comprise 92% of the Baptiste watershed and the remaining 8% drains directly to the lake by way of diffuse runoff areas (Figure 5). The total areas of individual stream basins and diffuse runoff areas are presented in Tables 2 and 3. Also included are the areas used to compute export coefficients for plant nutrients and other materials in subsequent sections.

Four major categories of land use are presently characteristic to the watershed: agricultural (cleared), forested (uncleared), wetlands (restricted drainage) and urban (primarily lakeshore dwellings) (Figure 6). Approximately 16 percent of the watershed area has been cleared for agricultural purposes with the main resource base being mixed farming. Fifty-eight percent is forested and belongs to the mixed wood section of the boreal forest region. The predominant species are black spruce (Picea mariana), trembling aspen (Populus tremuloides), balsam poplar (Populus balsamifera), willow (Salix sp.) and birch (Betula sp.). Twenty-five percent is covered by lakes, ponds and marshes and less than one percent consists of urban development (Figure 7).

Table 2. Stream basin areas (ha) and land use characteristics (% composition).

Stream Basin	Total	Excluding Urban and Standing Water	Excluding Urban, Wetland and Standing Water	Agriculture Area	Agriculture %	Forest Area	Forest %
A	893.4	893.4	848.6	724.0	85.3	124.6	14.7
B	83.9	83.9	61.8	57.5	93.0	4.3	7.0
C	283.7	283.7	268.3	162.7	60.0	105.6	39.4
D	733.3	728.8	661.0	173.9	26.3	487.0	73.7
E	9,050.1	8,981.5	7,391.9	1,283.7	17.4	6,108.2	82.6
F	5,656.8	5,455.8	3,269.6	54.9	1.7	3,214.7	98.3
H	86.3	86.3	66.8	20.2	30.2	46.6	69.8
I	357.4	357.4	357.4	112.7	31.5	244.7	68.5
J	111.4	111.4	111.4	31.4	28.2	80.0	71.8
K	308.9	308.9	247.1	17.8	7.2	229.3	92.8
L	8,901.4	8,839.0	6,629.2	1,092.0	16.5	5,537.2	83.5
M	70.5	70.5	62.4	61.8	99.0	0.6	1.0
N*	703.6	703.6	428.0	360.9	84.3	67.1	15.7
Total A-M	26,537.0	26,200.6	19,975.5	3,792.6		16.182.8	

N* - Tributary to outlet.



Rg.23
Rg.24
Rg.25
Tp.68
Tp.67
Athabasca R.
Island
Lake
Ghost Lake
Majors Lake
Baptiste
N
M
L
K
J
12
10
9

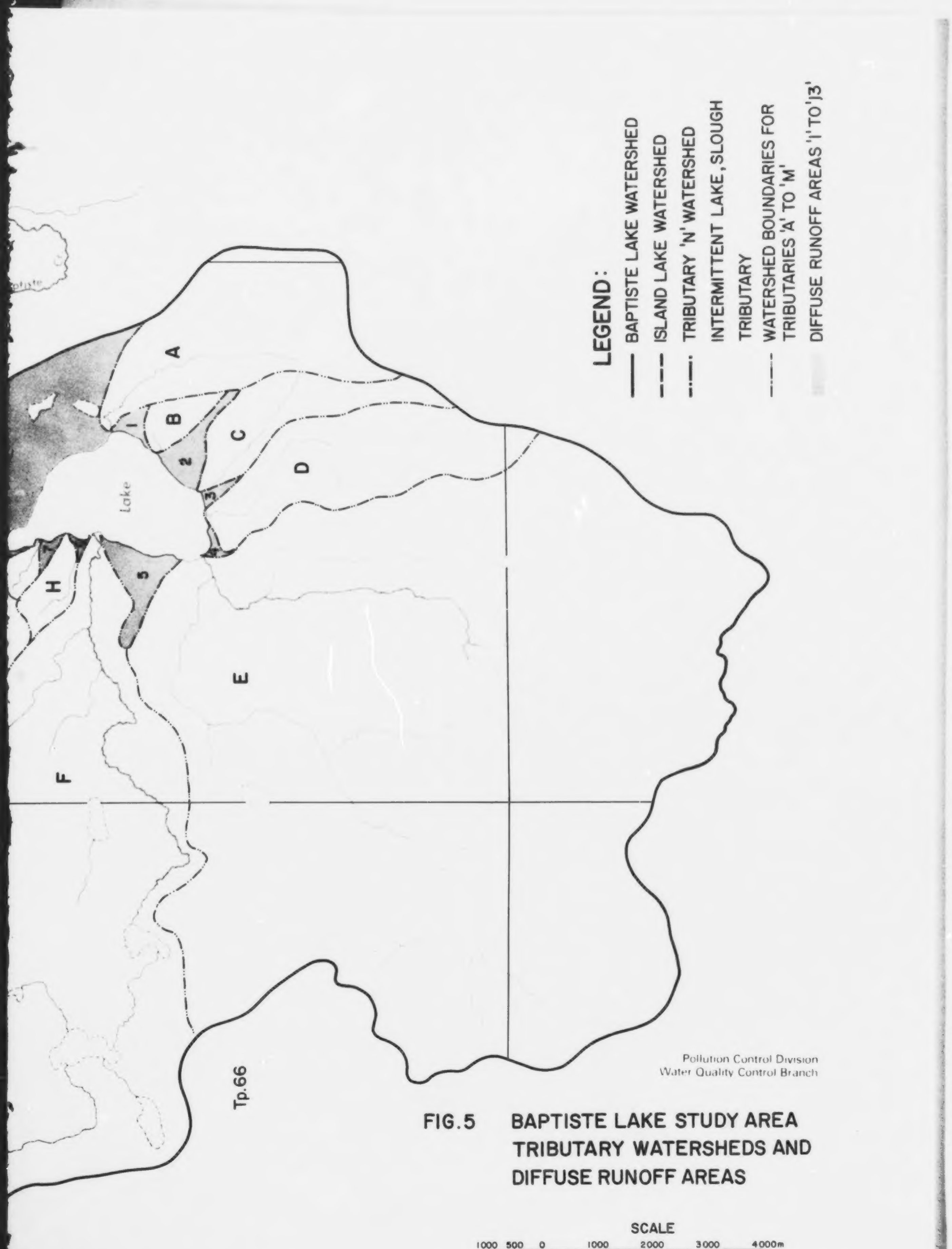

LEGEND:
BAPTISTE LAKE WATERSHED
ISLAND LAKE WATERSHED
TRIBUTARY 'N' WATERSHED
INTERMITTENT LAKE, SLOUGH
TRIBUTARY
WATERSHED BOUNDARIES FOR TRIBUTARIES 'A' TO 'M'
DIFFUSE RUNOFF AREAS '1' TO '13'
A
B
C
D
E
F
H
1
2
3
4
5
Lake
Tp.66
SCALE
1000 500 0 1000 2000 3000 4000m

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FIG.5 BAPTISTE LAKE STUDY AREA TRIBUTARY WATERSHEDS AND DIFFUSE RUNOFF AREAS

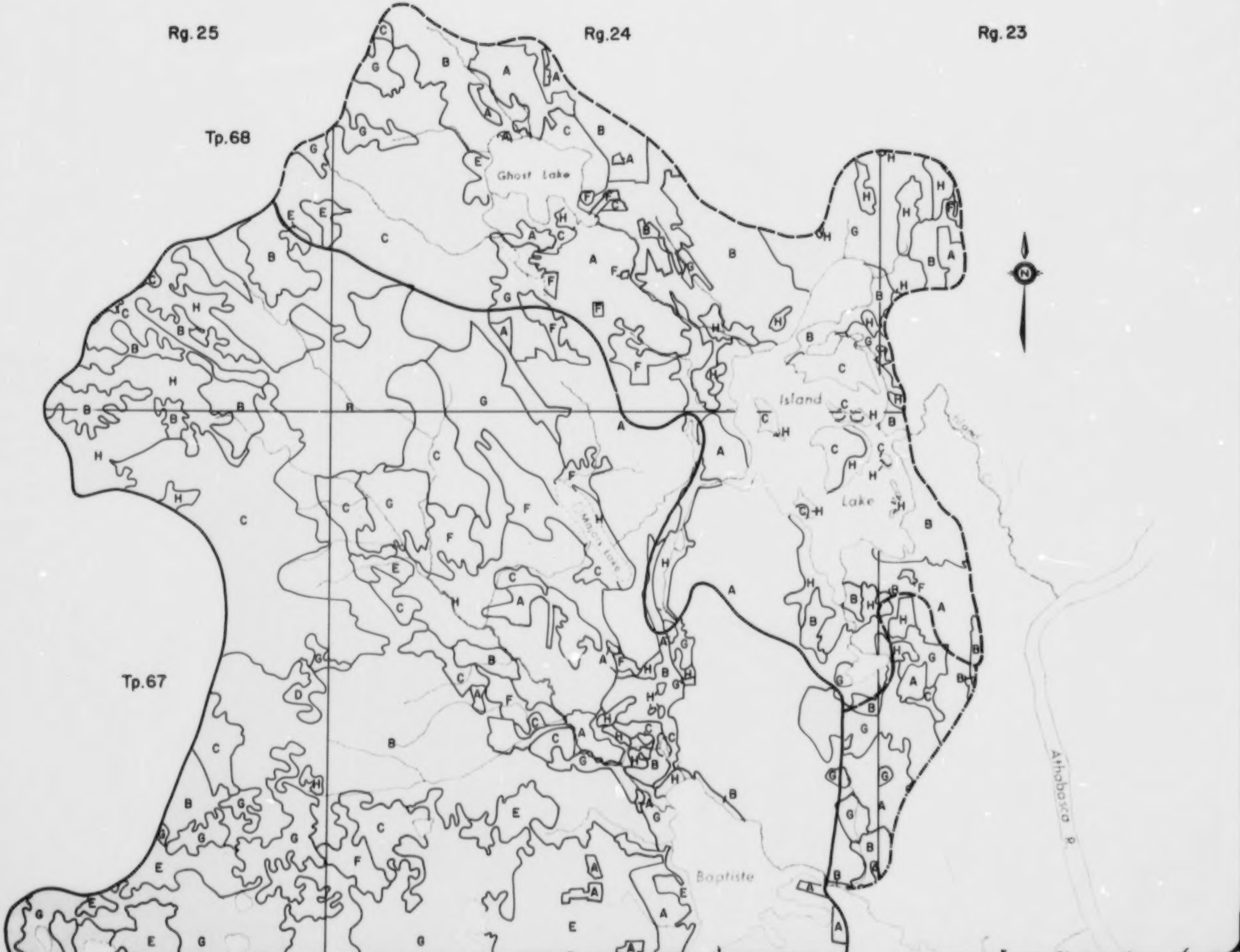
Table 3. Diffuse runoff areas (ha), and land use characteristics (% composition).

Diffuse Runoff Area	Total	Excluding Urban and Standing Water	Excluding Urban, Wetland and Standing Water	Agriculture Area	Agriculture %	Forest Area	Forest %
1	23.2	18.7	18.7	17.1	91.4	1.6	8.6
2	80.2	71.0	55.7	50.3	90.3	5.4	9.7
3	19.9	17.0	17.0	4.5	26.5	12.5	73.9
4	15.1	8.5	8.5	0	0	8.5	100.
5	108.5	103.5	100.9	38.2	37.9	62.7	62.1
6	16.2	15.1	10.0	2.2	22.0	7.8	78.0
7	13.4	11.1	11.1	11.1	100.0	0	0
8	57.8	47.4	47.4	38.9	82.1	8.5	17.9
9	234.6	230.9	214.3	53.9	25.1	160.4	74.9
10	35.4	35.4	23.8	0	0	23.8	100.
11	9.3	8.9	3.9	0	0	3.9	100
12	787.3	759.7	690.6	690.6	100.	0	0
13	838.6	802.2	710.3	123.4	17.4	586.9	82.6
Total	2,239.5	2,129.2	1,912.2	1,030.2		882.0	

The resident population of the entire watershed is not known, but is probably on the order of 100 people. During the summer months additional people occupy cottages around lakeshore. On approximately 500 lots there were 308 dwellings (293 cottages and 15 permanent residences) in 1976 and 325 (310 cottage and 15 permanent residences) in 1977 (Figure 8). Several backlot areas and lots within recent subdivisions are not yet developed.

A stream basin was considered to be either forested or agricultural if the percent of either land use type was equal to or greater than 85 percent (Dillon and Kirchner 1975). Percentages were estimated from a total basin area which excluded urban lands, wetlands, and areas of standing water (see Section 3.9).

Stream basins "E", "F", "K", and "L" were consequently defined as forested and "A", "B", "M", and "N" predominantly agricultural. Although "N" was a tributary basin to the outlet stream it was used in all land use/nutrient runoff analyses because of its high incidence of agricultural activity and close proximity to the study area.


Rg.25
Rg.24
Rg.23
Tp.68
Tp.67
Ghost Lake
Island Lake
Majors Lake
Baptiste
Athabasca R.
N

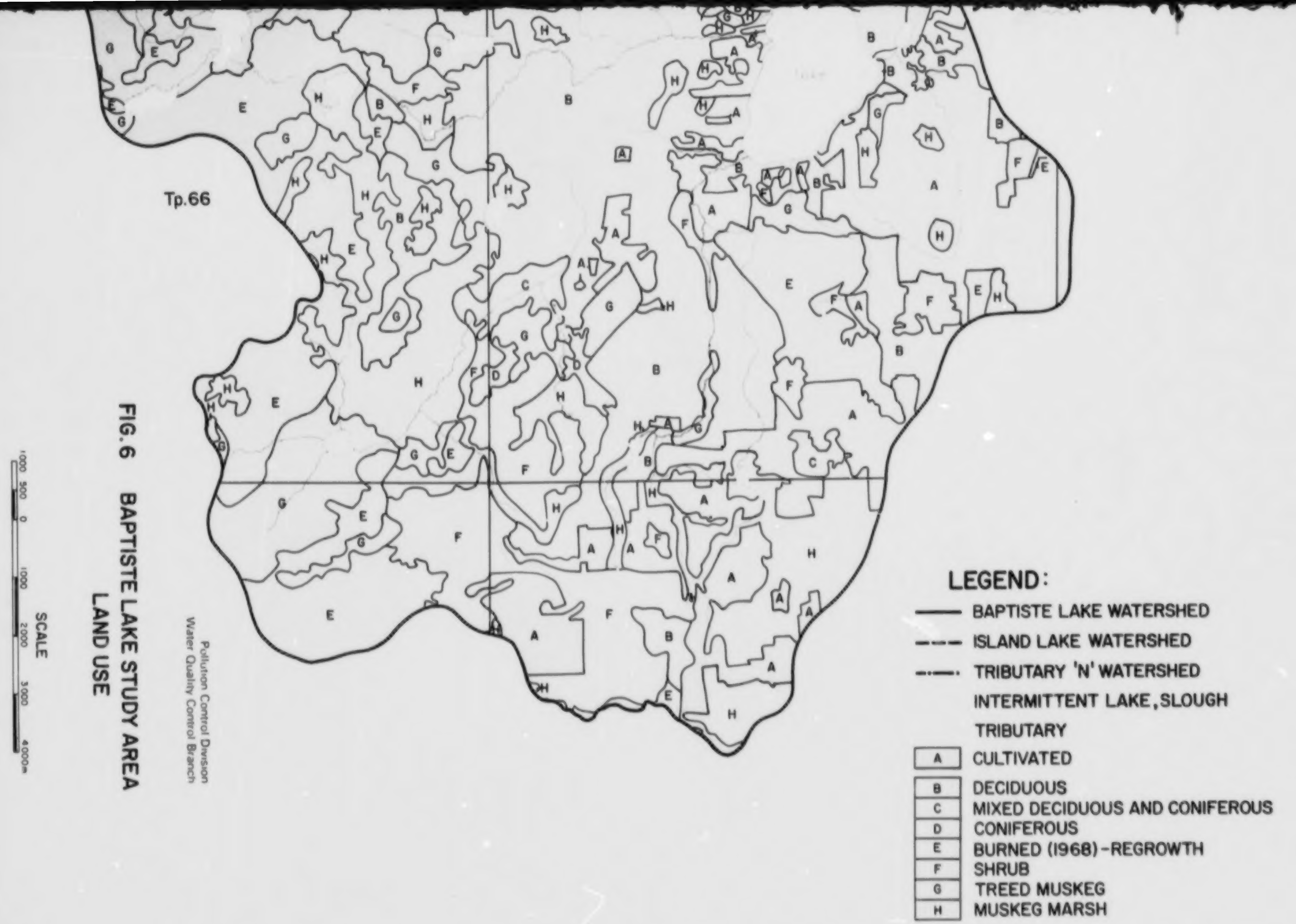

LEGEND:
BAPTISTE LAKE WATERSHED
ISLAND LAKE WATERSHED
TRIBUTARY 'N' WATERSHED
INTERMITTENT LAKE, SLOUGH
TRIBUTARY
A CULTIVATED
B DECIDUOUS
C MIXED DECIDUOUS AND CONIFEROUS
D CONIFEROUS
E BURNED (1968) - REGROWTH
F SHRUB
G TREED MUSKEG
H MUSKEG MARSH
Tp.66
SCALE
1000 500 0 1000 2000 3000 4000m

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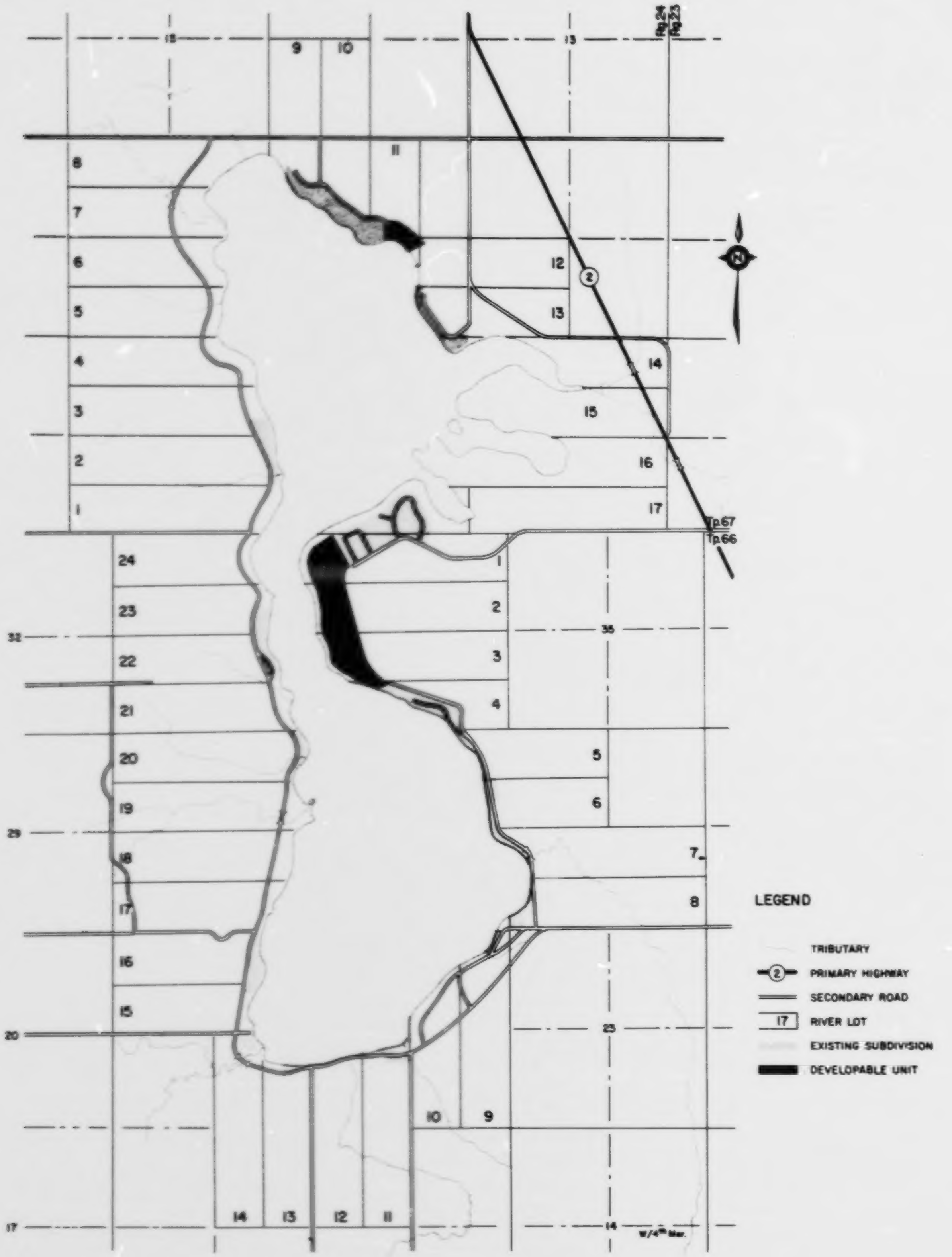
FIG. 6 BAPTISTE LAKE STUDY AREA
LAND USE


Rg24
Rg23
18
13
9
10
11
8
7
6
5
4
3
2
1
12
13
14
15
16
17
Tp67
Tp66
24
23
22
21
20
19
18
17
16
15
1
2
3
4
5
6
7
8
35
32
29
20
25
10
9
14
13
12
11
17
14
W/4th Mer.
LEGEND
TRIBUTARY
2
PRIMARY HIGHWAY
SECONDARY ROAD
17
RIVER LOT
EXISTING SUBDIVISION
DEVELOPABLE UNIT

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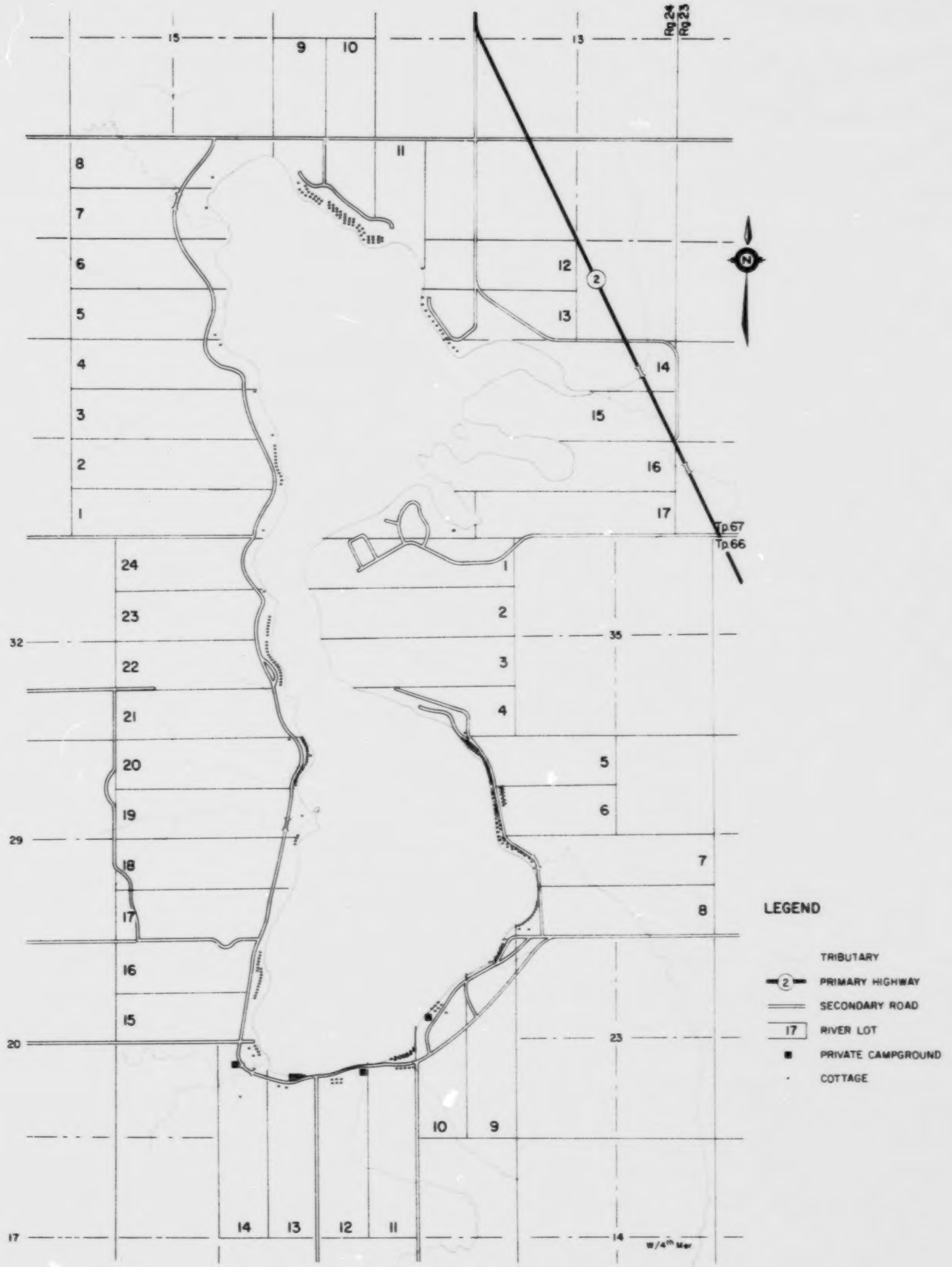
Water Quality Control Branch

FIG. 7 BAPTISTE LAKE STUDY AREA
SUBDIVISION MAP


Rg 24
Rg 23
15
9
10
13
11
8
7
6
5
4
3
2
1
12
13
14
15
16
17
2
Tp 67
Tp 66
N
24
23
22
21
20
19
18
17
16
15
32
29
20
17
1
2
3
4
35
5
6
7
8
23
10
9
14
13
12
11
14
W/4th Mer
LEGEND
TRIBUTARY
PRIMARY HIGHWAY
SECONDARY ROAD
17 RIVER LOT
■ PRIVATE CAMPGROUND
· COTTAGE

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Water Quality Control Branch

FIG. 8 BAPTISTE LAKE STUDY AREA
COTTAGE DISTRIBUTION (1976)

3.0 METHODS AND MATERIALS

3.1 SURFACE WATER HYDROLOGY

The water year from March 16th to March 15th of the following year was chosen for the Baptiste Lake watershed. Water samples and measurements of stream discharge were taken fortnightly from March to June 1976, monthly from July to December, 1976, and weekly from March, 1977 to March, 1979. More intensive sampling was undertaken during periods of peak flow. Stream sampling sites were located immediately adjacent to Baptiste Lake and were selected for their accessibility, uniformity of substrate, absence of weeds and obstructions, consistent flow pattern, and convenient sampling depths. In 1976 a float technique was used whereby the discharge rate was a product of surface velocity (corrected by a factor of 0.88) and stream profile area (Gray 1973). In 1977 and 1978 instantaneous stream velocities were measured with a GURLEY 622 current meter, and discharges were determined using procedures outlined in the Hydrometric Field Procedures Manual III (Water Survey of Canada).

Staff gauges were placed on inlets "D", "E", "F", and "L", stream N (tributary to outflow) and the outflow "O" in order to establish flow rating curves based on current meter flow and stage measurements (Figure 9). The rating curves were then used to estimate stream discharges when it was impossible to make manual measurements. A continuous, automatic, water stage recorder was installed on the largest inlet (stream "E") and the outflow, and information collected was used to derive daily mean stream levels and annual hydrographs. Annual hydraulic loads via tributary streams were estimated by integrating the measured daily loads of each stream over the annual cycle. Areal hydraulic yields were calculated by dividing the load by the total stream basin area. These yields were used to determine the theoretical discharge through diffuse runoff areas. An average areal yield for the watershed was determined by dividing the total volume of discharge from stream basins to the lake by the sum of basin areas. The percentage distribution of hydraulic loadings to each basin of the lake was estimated for each of the three study years. The total input and percentage contribution from all Baptiste inlets for the three years was also determined.

Long-term runoff volumes for the Baptiste Lake watershed from 1967 to 1978 were calculated using annual yields (extended) for the Dapp Creek watershed (Water Survey of Canada, Station 07BC006).

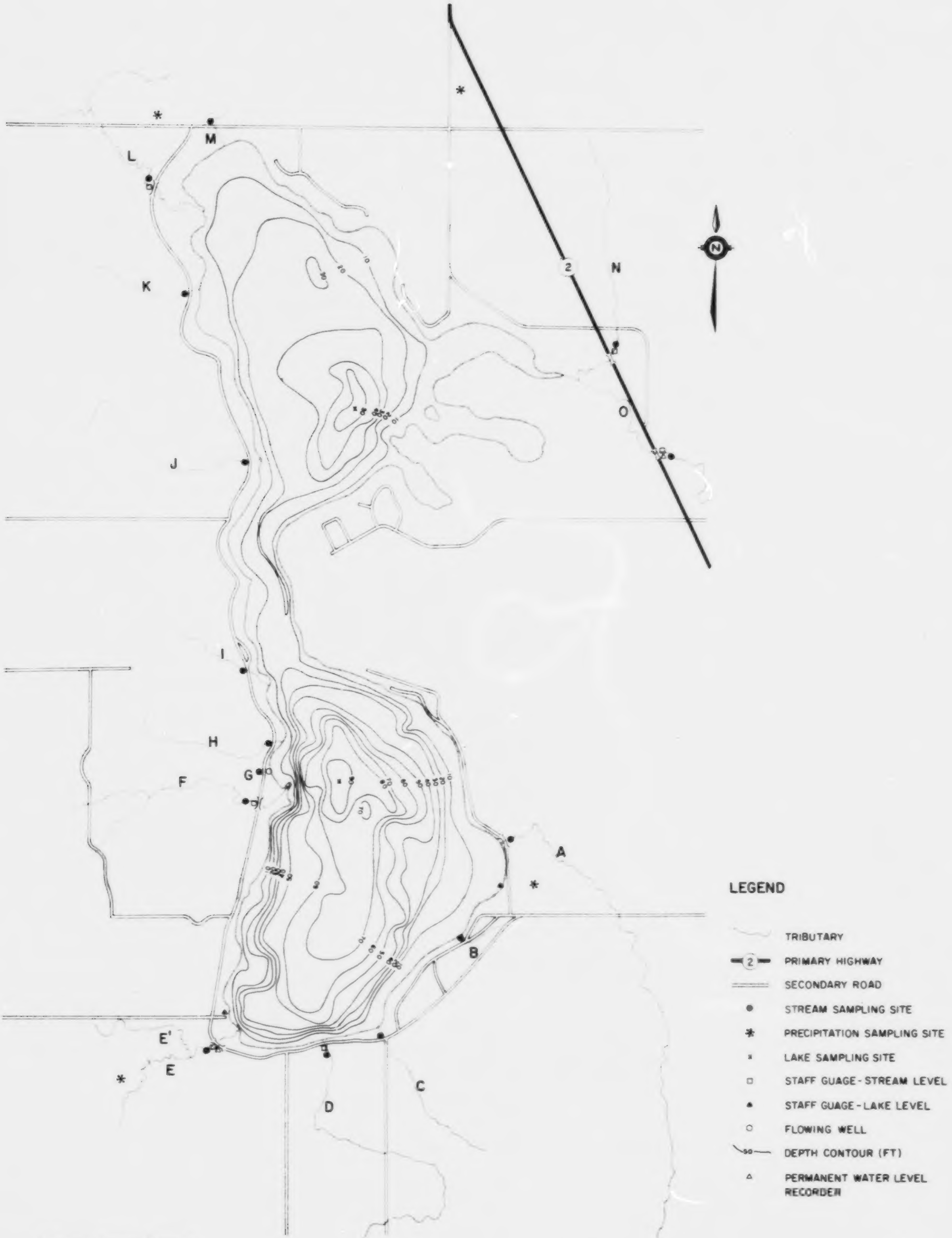
3.2 GROUNDWATER HYDROLOGY

The Earth Sciences Division of Alberta Environment was requested to assess the quantity and quality of groundwater interaction with Baptiste Lake. The initial phase of the work involved a field program to investigate the lithological and stratigraphic characteristics of the area and to investigate the groundwater flow to Baptiste Lake using the Darcy Equation (Davis and Dewiest 1966):

i.e. $Q = K I A$

where Q = volume of flow (m^3 day $^{-1}$)

K = hydraulic conductivity of material (m day $^{-1}$)


M
L
K
J
I
H
G
F
N
O
A
B
C
D
E'
E
2
LEGEND
TRIBUTARY
2 PRIMARY HIGHWAY
SECONDARY ROAD
● STREAM SAMPLING SITE
* PRECIPITATION SAMPLING SITE
x LAKE SAMPLING SITE
□ STAFF GUAGE - STREAM LEVEL
▪ STAFF GUAGE - LAKE LEVEL
○ FLOWING WELL
50 DEPTH CONTOUR (FT)
△ PERMANENT WATER LEVEL RECORDER

Pollution Control Division
Water Quality Control Branch

FIG. 9 BAPTISTE LAKE STUDY AREA
SAMPLING SITES

I = hydraulic gradient

A = cross-sectional area of material (m^2)

The second phase involved the development of a practical, lumped response watershed model to simulate hydrologic and hydrochemical transport with specific application to prairie lake-watershed systems (Crowe 1979). The model was designed to utilize basic hydrological, climatic and physical variables with a minimum of hydrogeochemical data determined in situ. The model operates by routing water through an idealized system of hydrostratigraphic units and is capable of simulating the hydrologic responses of a lake-watershed system to time variations in precipitation, potential evaporation and air temperatures. Model predictions are monthly and as the simulation moves forward in time, the movement of water in the various land-based components of the hydrological cycle and in the lake is quantified.

Mass transport is also modelled and consequently each major source of water that flows to the lake was chemically defined by total dissolved solids or ionic composition. The model predicts lake salinity and has been designed to account for processes such as ice formation and evaporation.

3.3 PRECIPITATION

Analysis of long-term precipitation for the Baptiste Lake area was based on the Baptiste Lake Station; when records were not available at this site, the data from station Athabasca 2 were used (Atmospheric Environment Service, Fisheries and Environment Canada).

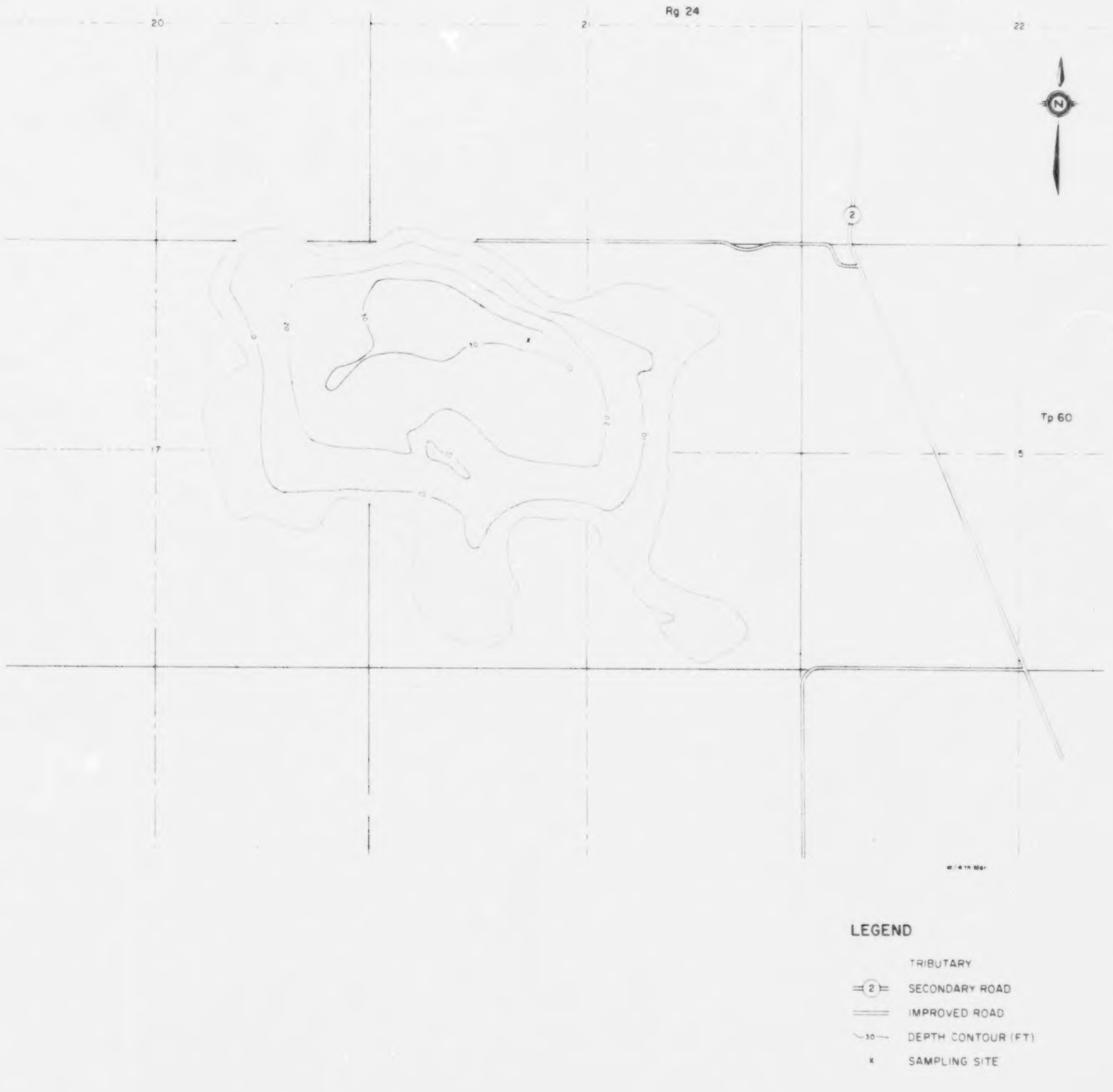
3.4 EVAPORATION

Lake evaporation for Baptiste Lake was calculated from pan evaporation measurements made at Edmonton International Airport, using regional evaporation adjustments outlined by the Saskatchewan Nelson Basin Board (1972). The SNBB adjustment to estimate Baptiste Lake evaporation from that of Edmonton is 0.882.

3.5 SAMPLING AND CHEMICAL ANALYSIS

Two principal sampling sites were established on Baptiste Lake (Figure 9). One was situated over the deepest part of the south basin in approximately 27.5 m of water, and the other over the deepest part of the north basin in aproximately 14.5 m of water. Similarly, the stations selected in Ghost (Figure 10) and Island (Figure 11) lakes were at their deepest points, 11.5 m and 18.5 m respectively. Water samples for phytoplankton and chemical analysis were collected in a 3 litre PVC Van Dorn bottle from the following depths: 0, 0.5, 1.0, 1.5, 2.0, 3.0, 6.0, and 10.0, and in the case of the south basin at 20.0 m. The sampling frequency was weekly during the ice-free season and fortnightly after freeze-up. Ghost and Island Lakes were sampled monthly. Stream sampling sites and frequencies were similar to those established for discharge measurements.

Water samples for chemical analysis were cooled and transported to the laboratory in 4 oz. and 32 oz. polyethylene bottles; field filtering was not performed on any samples. Analyses were carried out at the former Pollution Control Division Laboratory of Alberta Environment in Edmonton (now Alberta Environmental


Rg 24
20
21
22
N
2
Tp 60
17
5
10
20
LEGEND
TRIBUTARY
2 SECONDARY ROAD
IMPROVED ROAD
30 DEPTH CONTOUR (FT)
x SAMPLING SITE

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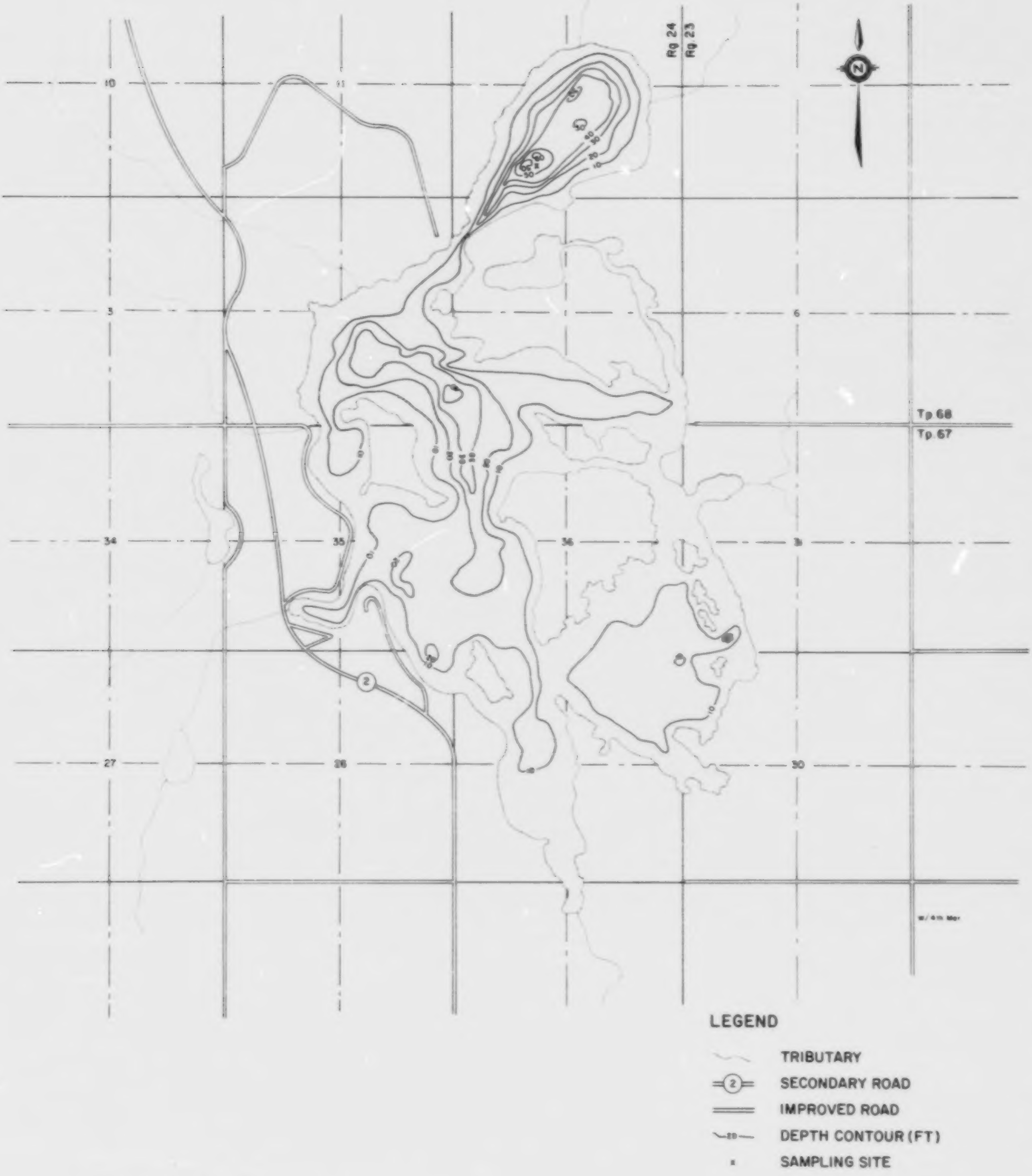
Water Quality Control Branch

FIG. 10 GHOST LAKE SHOWING BATHYMETRIC CONTOURS AND SAMPLING SITE



Rg 24
Rg 23
N
10
11
3
6
Tp 68
Tp 67
34
35
36
31
27
28
30
2
50
40
30
20
10
LEGEND
TRIBUTARY
SECONDARY ROAD
IMPROVED ROAD
DEPTH CONTOUR (FT)
SAMPLING SITE

Pollution Control Division
Water Quality Control Branch

FIG. 11 ISLAND LAKE SHOWING BATHYMETRIC CONTOURS AND SAMPLING SITE

Center, Vegreville). The methods utilized are summarized by their respective NAQUADAT code numbers in Table 4. During 1976 and 1977 a number of stream samples were frozen prior to analysis due to logistical constraints in the laboratory. The results of the analyses on frozen samples have been included in the loading estimates and have been discussed in the appropriate sections.

3.6 PHYSICAL ANALYSIS

Vertical profiles of temperature, dissolved oxygen and conductivity were recorded with Hydrolab Surveyor model 60 in situ water quality analyzer. Extinction of photosynthetically available radiation (taken as 400-700 nm) was measured with Kahlsico Universal Radiometric Submarine Photometer.

Light extinction coefficients have been calculated as:

$$E = \frac{\ln Io - \ln I}{z}$$

where "I_o" is the initial light intensity and "I" the intensity after passage through a water column z meters thick. The extinction coefficients (ln units m^{-1}) were calculated between 1 - 2 m and 2 - 3 m in order to coincide with measurements of standing crop and to avoid inaccuracies caused by surface turbulence.

Total surface irradiance during primary production experiments was measured on a Belfort Instrument Company Actinograph. Total irradiance data was also extracted from the monthly Radiation Summary, Environment Canada for Edmonton Stony Plain (58° 33'N; 114° 06'W) and Beaverlodge CDA (55° 12'N; 119° 25'W).

Table 4. Chemical analysis methods.

Parameter	Naquadat Code	Detection Limit
Calcium	20105 L	1 mg L^{-1} as Ca
Magnesium	12102 L	1 mg L^{-1} as Mg
Sodium	11103 L	1 mg L^{-1} as Na
Potassium	19103 L	0.02 mg L^{-1} as K
Chloride	17203 L	1 mg L^{-1} as Cl
Sulphate	16306 L	1.0 mg L^{-1} as SO_4
Total Alkalinity	10101 L	0.5 mg L^{-1} as $CaCO_3$
pH	10301 L	0.01 pH units
Bicarbonate	06201 L	0.5 mg L^{-1} as HCO_3
Total Hardness	10604 L	5 mg L^{-1} as $CaCO_3$
Silica	14102 L	0.5 mg L^{-1} as SiO_2
Conductance	02041 L	0.2 μhos cm^{-1}
Total Organic Carbon	06001 L	1 mg L^{-1} as C
Total Inorganic Carbon	06051 L	1 mg L^{-1} as C
Total Carbon	06006 L	1 mg L^{-1} as C
NO_2 + NO_3 Nitrogen	07111 L	0.002 mg L^{-1} as N
Ammonia Nitrogen	07555 L (to 15/07/76)	0.2 mg L^{-1} as N
Ammonia Nitrogen	07555 L (to 15/11/77)	0.05 mg L^{-1} as N
Ammonia Nitrogen	07561 L (from 24/3/78)	0.05 mg L^{-1} as N
Total Dissolved Solids	00202 L	
Total Kjeldhal Nitrogen	07003 L (to 15/11/77)	0.05 mg L^{-1} as N
Total Kjeldhal Nitrogen	07015 L (from 24/3/78)	0.05 mg L^{-1} as N
Total Phosphorus	15001 L (from 15/11/77)	0.05 mg L^{-1} as P
Total Phosphorus	15419 L (from 24/3/78)	0.05 mg L^{-1} as P
Orthophosphate (SRP)	15256 L	0.002 mg L^{-1} as P
Copper	29305 L	0.001 mg L^{-1} as Cu
Iron	26302 L	0.01 mg L^{-1} as Fe
Manganese	25304 L	0.008 mg L^{-1} as Mn
Zinc	30305 L	0.001 mg L^{-1} as Zn
Vanadium	23303 L	0.002 mg L^{-1} as V
Cobalt	27302 L	0.001 mg L^{-1} as Co
Molybdenum	42304 L	0.001 mg L^{-1} as Mo

3.7 TROPHIC ASSESSMENT

Phytoplankton samples collected for enumeration and identification were stored in 4 oz. Wheaton snap cap jars and preserved with 5 mL of acid Lugol's iodine solution (a mixture of KI and I_2 in acetic acid). Subsamples (10 mL or appropriate dilutions) of these bulk water samples were sedimented for 24 hours in 25 mm outside diameter sedimentation chambers (Lund et al. 1958). Samples were counted on a horizontal transect of the chamber's diameter at 400x magnification using a Wild M-40 inverted microscope. Algal identifications were achieved through the use of the following taxonomic keys: Nygaard (1977); Patrick and Reimer (1966); Prescott (1970); Smith (1950); Stein (1975); Taft and Taft (1971); and Webber (1971). Cell numbers are based on individual cells as opposed to whole colonies or filaments. Cell volumes were estimated by equating cells to known geometric shapes and/or utilizing the volume estimates of other workers (McCart et al. 1979; Tsui et al. 1978; Bellinger 1974; Javornicky and Komarkova 1973; Hallegraeff 1976; Hallegraeff 1977). For some diatoms with irregular shapes valve area was determined utilizing exact drawings on graph paper from which either planimetric or arithmetic determinations were obtained. A uniform girdle width was then assumed to allow volume computations on a rectangular basis.

Water samples for chlorphyll a determinations were filtered onto glass fibre filters at 7.5 psi. The filters were sprinkled with magnesium carbonate, wrapped in tin foil, and frozen; extractions were subsequently conducted in 90 percent buffered acetone for 24 hours. Chlorophyll a was determined by the fluorometric methods of Yentsch and Menzel (1963) as modified by Holm-Hansen et al. (1965) using a Turner model III fluorometer calibrated against the trichromatic equations of Strickland and Parsons (1972). The instrument was equipped with a 5-60 excitation filter and a 2-64 transmission filter. Pigment concentrations were calculated using the following equations:

$$\text{Chlorophyll } \underline{a} = K \cdot (Rb - Ra) \cdot \frac{E}{V} \cdot \frac{1}{d} \text{ ug L}^{-1}$$

$$\text{Phaeophytin } \underline{a} = K \cdot (H \cdot Ra - Rb) \cdot \frac{E}{V} \cdot \frac{1}{d} \text{ ug L}^{-1}$$

$$\text{Acid Ratio} = Rb/Ra$$

where:

Rb = Pre-acidification fluorometer reading, corrected for blank.

Ra = Post-acidification fluorometer reading, corrected for blank.

E = Extraction volume in mL

V = Sample volume in litres

H = Acid ratio of pure Chlorophyll a in 90% acetone, determined as 1.92 on this instrument.

$K = F_D \frac{(H)}{H-1}$

F_D = Door factor, relating units of Chlorophyll a to units of fluorescence

d = Dilution factor, expressed as a decimal.

Phytoplankton primary production was estimated using the radiocarbon uptake method of Steeman Nielsen (1952) as modified by Schindler et al. (1973). The incubations were carried out in transparent 212 mL PVC bottles inoculated with 2.5 microcuries of

^{14}C-bicarbonate. Two light bottles and one dark bottle were incubated for each depth. Soeder and Talling in Vollenwieder (1971) have discussed the use of plastic materials in primary production work, and Hillman (1975) has evaluated the optical characteristics of PVC in relation to other plastics for the same purpose. Plastic bottles were used because of their logistical convenience and low costs (Tett, Wood, and Edwards 1973). Although quartz bottles are optically ideal for measurements of phytoplankton photosynthesis, they are very expensive. Most workers have used borosilicate glass bottles for incubations, however, the absorption of shorter wavelength light by glass is substantially higher than is the case either in natural waters or PVC. This feature may reduce light inhibition on surface photosynthetic measurements, therefore causing overestimates in integral photosynthesis.

In situ incubations were conducted from late morning to mid-afternoon and covered a time span of between 2 1/2 to 4 hours depending on weather conditions. At the end of the incubation period all samples were fixed with formalin to give a final solution strength of 5%. The acidification and bubbling step was carried out in the incubation bottles by adding 0.5 mL of 1 N HCL and introducing a stream of air at 15 psi through a disposable Pasteur pipet. The optimal bubbling time was empirically determined to be 30 minutes from controlled degassing experiments. We had previously observed algal material adhering to the side walls of the glass filtering vessels used by Schindler et al. (1973), as also noted by Theodorrson and Bjarnason (1975), which would have caused an underestimate of ^{14}C - uptake. Bubbling in the PVC incubation bottles permitted the capping and agitation of the contents after processing to ensure more complete re-suspension of any attached particulate material. A sub-sample of 5 mL was added to 10 mL of Aquasol (New England Nuclear) and counted in gel phase on a Nuclear-Chicago Unilux II liquid scintillation counter. Quench correction was effected by the channels ratio method, the efficiency curve being defined as:

$$y = .697x + .916 \qquad (r^2 = 0.97)$$

where y = efficiency

x = channels ratio

Single vertical haul samples for seston analysis were collected with a 80 um x 80 um micron mesh Wisconsin net from the total water column at each station and preserved with formalin. The samples were subsequently split for ash-free dry weight determination (particulate organic seston) according to the methods of Winberg (1971), and for zooplankton identification. Large Chaoborus sp. larvae were removed from samples prior to weighing. The sample was filtered onto a pre-weighed GFC filter paper, oven dried at 75°C for approximately 16 hours, and then ashed in pre-weighed crucibles for one hour at 500°C. The ash was subsequently rewetted with distilled water and brought to equilibrium in a drying oven at 75°C for 6 - 12 hours prior to final weighing.

3.8 STREAM CHEMICAL LOADINGS

Daily loadings via tributary streams were determined by combining concentrations with daily discharge measurements. Annual chemical loadings were estimated for three years by integrating the daily loadings through each stream over the annual cycle. The annual export of each stream was then converted to an annual export coefficient (kg ha^{-1} yr^{-1}) by incorporating drainage area into the calculation. The net contribution from wetlands was considered to be zero (Sonzogni and Lee 1974; Uttormark et al. 1974; Lee et al. 1975; Rast and Lee 1978) in all loadings and export coefficient estimates of total phosphorus, orthophosphate (SRP), total nitrogen, organic nitrogen, ammonia, and nitrate+nitrite. However, other workers have indicated that wetlands or muskeg areas can release considerable quantities of chemicals other than nutrients (Akena 1978; Lee, Bentley and Amundson 1975). Consequently, these areas have been included in the calculation of non-nutrient export coefficients. Urban lands have been excluded from the calculation of all export coefficients, and loadings from these areas are treated separately in the section dealing with shoreline development. Finally, the areas of standing water (ponds, small lakes, sloughs, etc.) have been excluded from all calculations since material retention in these waterbodies was impossible to determine.

In order to predict nutrient export from diffuse runoff areas (calculated for six nitrogen and phosphorus species), it was necessary to extrapolate from observed stream basin values. Diffuse runoff area values were estimated by selecting the coefficient (or average of two coefficients) of the stream basin(s) in which the relative proportion of land use types was most similar to the diffuse runoff area in question.

The percent distribution of annual nutrient loadings via surface runoff to each basin of Baptiste Lake, as well as monthly inputs and percentages, was estimated for each water year. Long-term total phosphorus and total nitrogen inputs were computed from 1968 to 1978 by multiplying the three year average flow-weighted mean concentration of total phosphorus and total nitrogen in tributary streams (1976-1978) with the estimated annual inflows (1968 to 1978) to Baptiste Lake. These inputs were then plotted on frequency curves in order to enable the prediction of 1:25 year minimum, maximum and long-term average loadings.

Daily (for all inputs combined) and annual (for individual streams) nitrogen:phosphorus mass ratios were estimated using both total and "available" loadings of the two nutrients (Rast and Lee 1978). The daily nutrient loadings from stream "G" (continuous groundwater flow through a small pipe directly to the lake) were negligible relative to the sum from all other inputs, although the daily N:P mass ratio estimates for this input were very high. Consequently, they were unrepresentative of ratios from regular surface runoff and have been omitted entirely from the daily N:P mass ratio estimates.

Annual loading values were used to estimate the ratios of orthophosphorus (SRP) to total phosphorus, and total inorganic nitrogen to total nitrogen for specific stream basins. As a result, we were able to compare the potential loss of "available" nutrients from different land use types.

An alternative method of deriving nutrient inputs via surface runoff is to calculate theoretical loadings by using nutrient export coefficients obtained from the literature. We selected the coefficients presented by Rast and Lee (1978) which were applied to both stream basins and diffuse runoff areas.

3.9 ATMOSPHERIC DEPOSITION OF NUTRIENTS

Nutrient input via precipitation and dustfall was estimated using two different approaches. From April to November of 1978, input was measured in 200 mm diameter bulk precipitation collectors which trap rainfall and dustfall and prevent evaporation (Likens et al. 1977). For the 1978-79 winter season, the input was estimated as the nutrient mass which acculated in 450 mm diameter plastic cans (Likens et al. 1977).

Both summer and winter collectors were left out for two to four weeks at a time and were placed at four locations around the lake in order to provide a representative estimate of deposition in regions of different types of land use and exposure. Input from April, 1978 to November, 1978 was determined by taking the mean deposition in the four collectors for each sampling period and totalling these periodic means. The 1978-79 winter input was determined by bulking all samples at each site, subsampling for nutrient content, and then averaging the loadings from all sites. The volume-weighted mean and range of nutrient concentrations from April, 1978 to November, 1978 and the winter period of November, 1978 to April, 1979 were estimated. In order to derive an approximate figure for annual nutrient input via atmospheric deposition, the observed rates from April, 1978 to April, 1979 were totalled.

3.10 SHORELINE DEVELOPMENT

The nutrient input associated with residential shoreline development on lakes arises from two sources: domestic wastes and surface runoff across cleared lots and roads. The potential transport of nitrogen and phosphorus into surface waters from sewage and wastewater discharge is dependent both upon methods of disposal and soil properties.

Phosphorus retention efficiencies range from nearly 100% in clay-loam soils where phosphorus is adsorbed quickly, to almost nil in sandy soils. Nitrogen, however, exhibits significantly higher rates of mobility through a variety of soils (Uttormark et al. 1974; Jones and Lee 1977). Although most lakeshore dwellings at Baptiste are built on a clay-till base, some are situated on sandy deposits thus creating a variety of seepage patterns. Because of these variabilities in soil structure and disposal methods it was viewed as impractical to measure the seepage from each point of discharge. Instead, the maximum inputs of nitrogen and phosphorus that could possibly arise as a result of the existing and projected levels of development were estimated. This approach deliberately overestimates the loadings from shoreline dwellings and presents them in their worst perspective.

Following the method of Dillon and Rigler (1975), total phosphorus and total nitrogen loadings from shoreline dwellings can be calculated as follows:

$S = W \times N \times T$

where S = Annual input (kg yr^{-1})

W = Theoretical annual per capita output (kg capita^{-1} yr^{-1})

N = Number of dwellings

T = Mean number of capita-years at each dwelling per year.

Numerous estimates of the annual per capita outputs of total phosphorus and total nitrogen have been made from the chemical analysis of municipal effluents; these values depend on the relative contributions of human excrement, household waste, detergents and storm runoff to the sewage (Dillon and Rigler 1975; Vollenweider 1968). In order that the figures utilized were representative of local conditions, the mean per capita through-puts of total nitrogen and total phosphorus for the town of Athabasca, Alberta, were used to define "W" for Baptiste Lake. These data were obtained from the Muncipal Engineering Branch, Pollution Control Division, Alberta Environment (1976, 1977(a)). "W" was calculated as the total yearly through-put of each nutrient divided by the population of Athabasca for the same year.

A questionnaire distributed in 1976 provided information on recreational user pressure for Baptiste Lake (Alberta Municipal Affairs 1977). In order to incorporate the different values for "T", the Dillon and Rigler (1975) equation was modified as follows:

$S = W[(Nc \times Tc) + (Np \times Tp)]$

Nc = number of lakeshore cottages

Tc = mean number of capita years spent at each cottage per year

Np = number of permanent lakeshore dwellings

Tp = mean number of capita years spent at each permanent dwelling per year.

In order to assess the nutrient contribution from surface runoff over cleared lakeshore lots, the nutrient export coefficient of Rast and Lee (1978) for urban lands was applied to the total area of lots surrounding the lake. In order to estimate the potential urban runoff in the remaining sub-divided lots and from those undeveloped lots in phase 2 of the Whispering Hills development, lot areas were extracted from subdivision plans or assumed to be 100 ft. x 200 ft. The existing and projected nutrient loads arising from the various development scenarios were determined.

3.11 LEAF LITTER NUTRIENT LOADING

In order to estimate the input of nutrients to Baptiste Lake through leaf litter, a cluster of three leaf collectors was placed at each of four sites on the lake shoreline from September 29, 1978 to October 13, 1978. Collectors consisted of styrofoam trays (1 meter square) with plastic mesh sides (50 cm high). A cement block was used to anchor the collectors. They were placed approximately three meters apart perpendicular to the shoreline and sites were selected where the composition of the vegetation was representative of the shoreline. Leaves from each collector were separated according to species, weighed, stored in plastic bags and sent to the lab for nutrient analysis.

From each collector and leaf type, a one gram subsample was taken, weighed, heated in an oven for one hour at 90°C, weighed again, ground up, rinsed with deionized water and transferred to a blender. The volume was brought up to 500 mL and blended for five minutes at the highest speed. The subsample was transferred to a plastic bottle, preserved with acid and stored at 4°C. These subsamples were later analyzed for ammonia, total Kjeldhal nitrogen (TKN), total phosphorus, soluble reactive phosphorus, and low level nitrite concentrations. Interferences made it impossible to run low level nitrate. Total nitrogen was calculated as the sum of TKN and nitrite.

All results from the subsample analyses were extrapolated to the total samples to determine nutrient input from each leaf type. The sum from leaf types made up the total input from each collector. By intregrating the loading from the three collectors at a particular site over a nine metre distance (the distance from the shoreline to the outer edge of the last collector) and multiplying by either the total length of wooded lake shoreline, or the length of wooded shoreline which could potentially contribute due to the effects of prevailing winds (N.W.), we were able to estimate crudely the loadings of each nutrient to the lake from this source.

4.0 CLIMATE

The Baptiste Lake watershed is subject to a continental climate, winters are long and cold, the summers are short but warm, and the humidity is low. The mean daily temperatures, the mean daily maximum temperatures, and the mean daily minimum temperatures for Athabasca 2 are summarized in Table 5 (Environment Canada 1981).

The area receives approximate one-third of its annual precipitation during July and August. The total monthly precipitation at Baptiste Lake is illustrated in Figure 12 and the longer record of Athabasca 2 (Atmospheric Environment Service, Fisheries and Environment Canada) in Figure 13. The long-term annual precipitation (1953-1978) for Athabasca is 19.40 inches (49.27 cm) and the long-term trends are illustrated in Figure 14. During the period of study on the lake, 1976 was characterized as a low precipitation year (17.84 inches), and 1977 (23.38 inches) as a high precipitation year. The third year of work carried out on the streams was also a high precipitation period (24.25 inches).

The average frost-free period at Baptiste Lake is 80-100 days. The last frost in spring normally occurs approximately June 1 - 15; the first frost in autumn usually occurs between September 1 - 15 (Atlas of Alberta).

The area receives approximately 2100 - 2200 hours of bright sunlight per year (based on measurements during the period of 1931-60) and the mean daily global solar radiation values are summarized in Table 6 (Atlas of Climatic Maps, Canada Department of Transport 1968).

Wind speed and direction were not monitored at Baptiste Lake; however, information is available from the Edmonton Municipal Airport. Southerly and northwesterly winds dominate the annual weather. Northerly and northwesterly winds tend to be the strongest, averaging 11 - 13 m.p.h. During the summer months, winds are commonly from the west, northwest, and south, between 9 and 11 m.p.h.

Lake ice generally forms in mid-November and disappears at the beginning of May. Spring runoff occurs during April, and stream ice also forms during mid-November.

Table 5. Monthly breakdown of temperature (°C) at Athabasca 2.

Month	Mean Daily Temperature	Mean Daily Max. Temperature	Mean Daily Min. Temperature
Jan	-17.9	-12.9	-22.9
Feb	-11.9	- 6.3	-17.3
Mar	- 6.4	- 0.5	-12.3
Apr	3.4	9.5	- 2.8
May	10.1	16.7	3.4
Jun	14.1	20.4	7.8
Jul	16.2	22.5	10.0
Aug	14.8	20.8	8.8
Sep	9.5	15.2	3.9
Oct	4.6	9.9	- 0.7
Nov	- 6.1	- 1.9	-10.3
Dec	-13.8	- 9.2	-18.4
Year	1.4	7.0	- 4.2

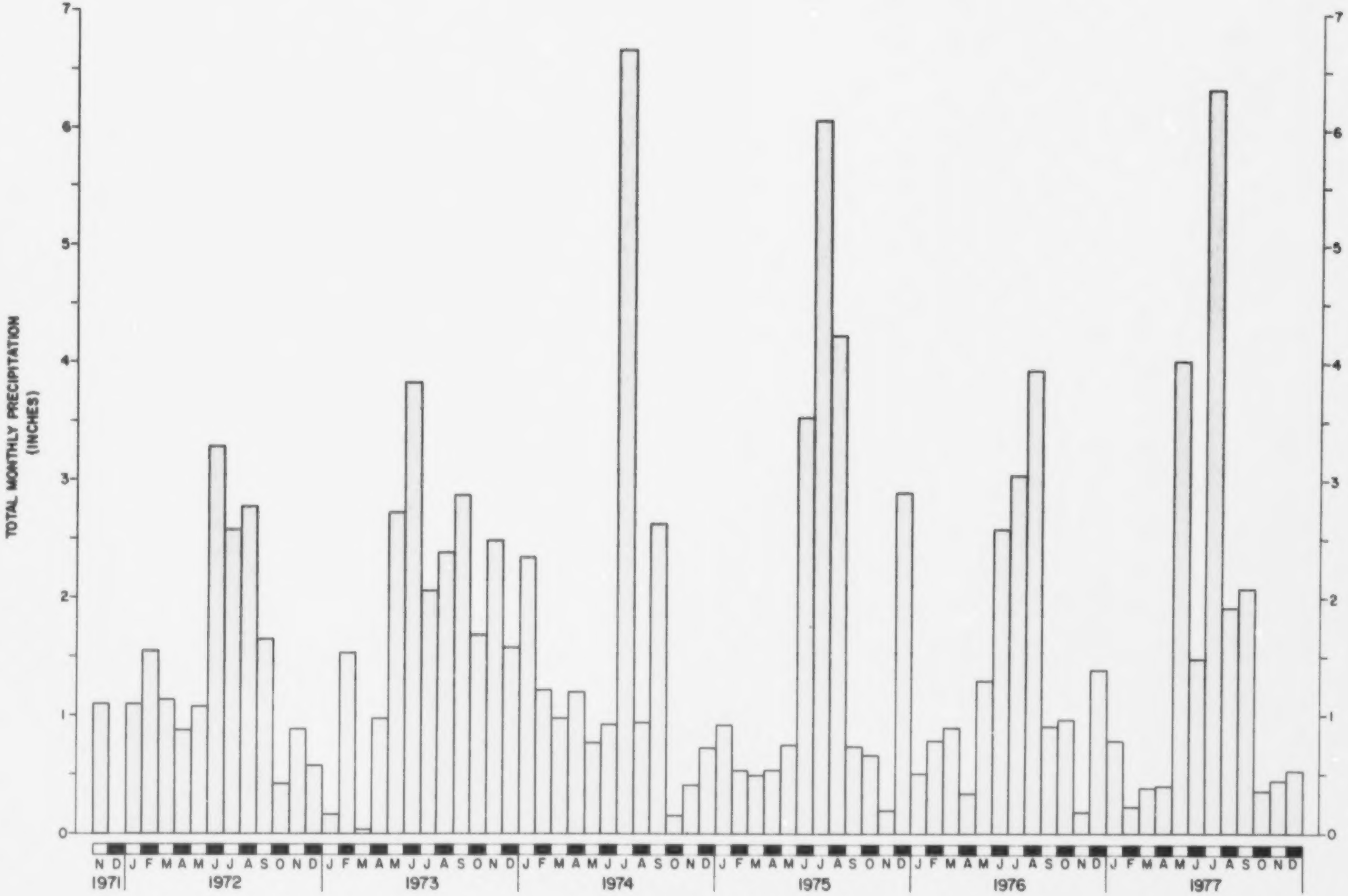


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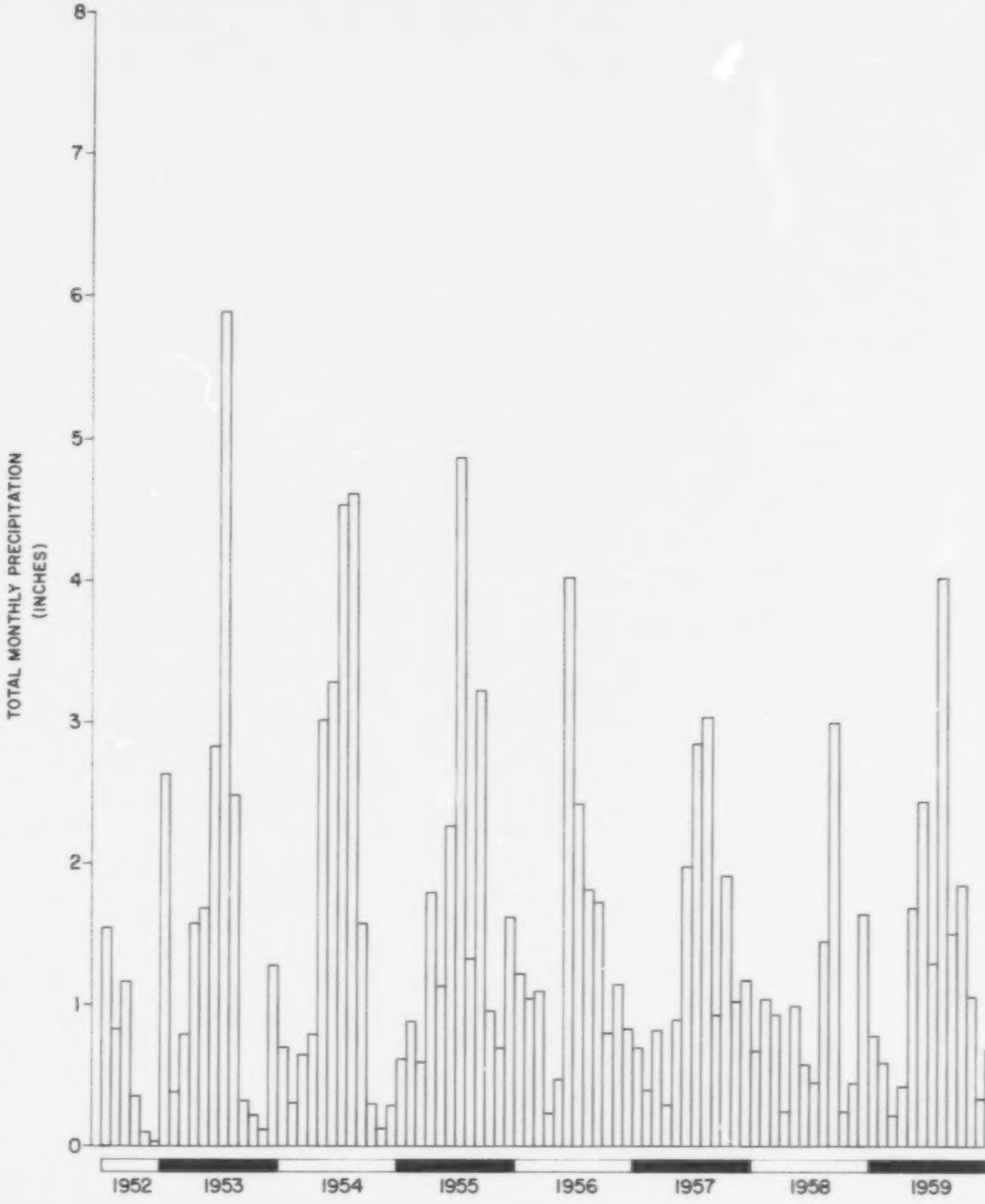
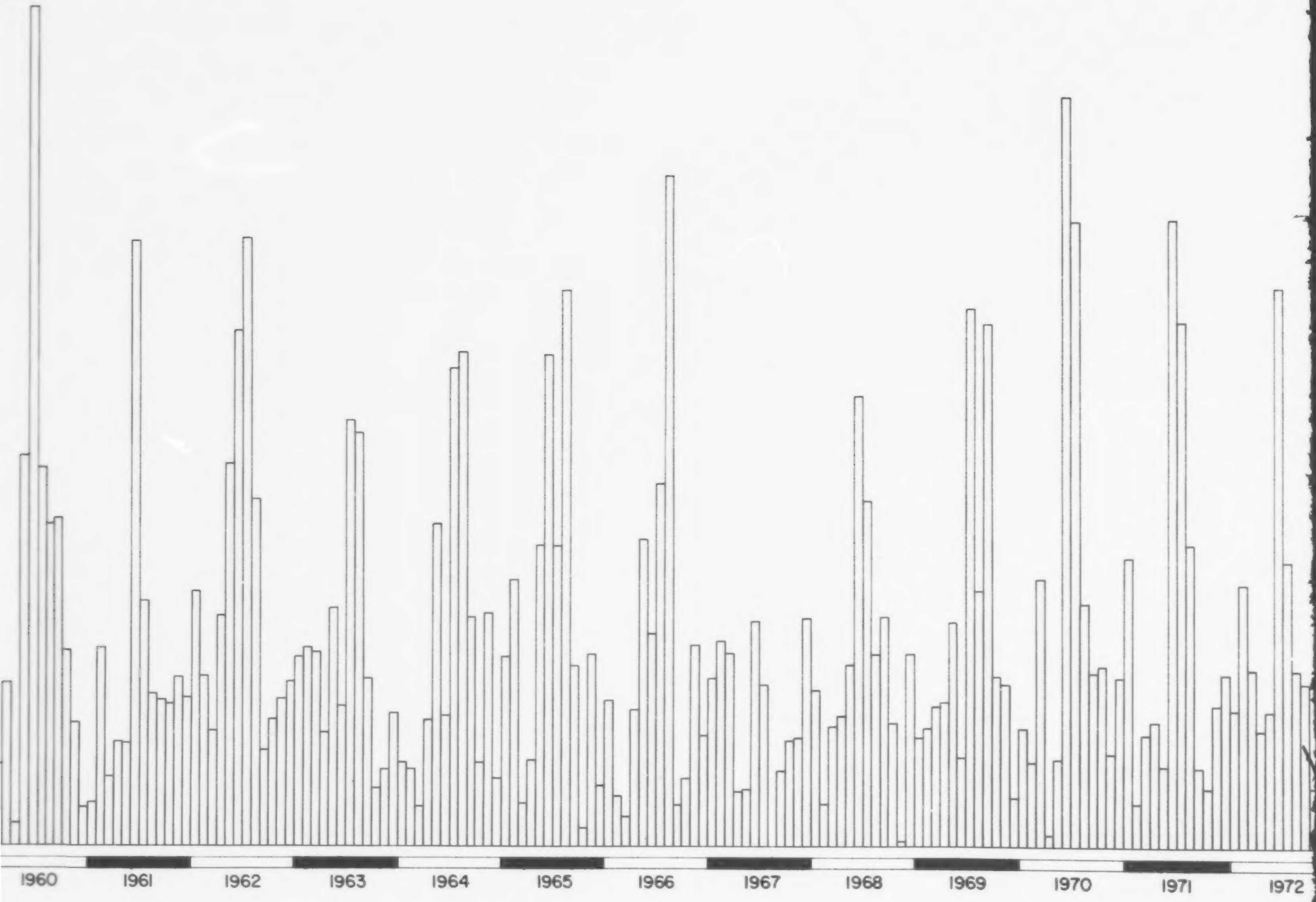
FIG.12 TOTAL MONTHLY PRECIPITATION AT BAPTISTE LAKE


TOTAL MONTHLY PRECIPITATION
(INCHES)
8
7
6
5
4
3
2
1
0
1952
1953
1954
1955
1956
1957
1958
1959



Pollution Control Division
Water Quality Control Branch

FIG. 13 TOTAL

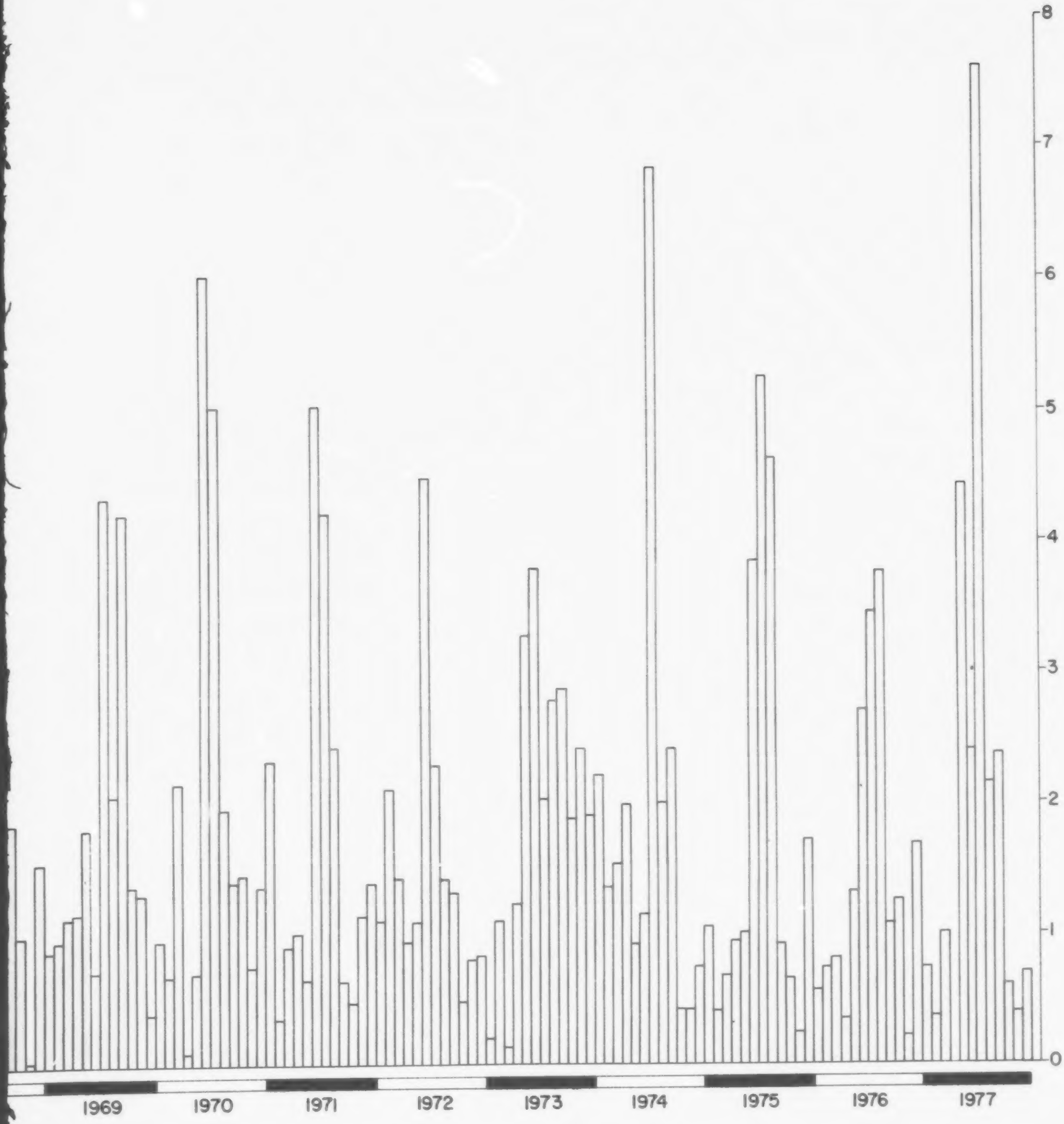


FIG. 13 TOTAL MONTHLY PRECIPITATION AT ATHABASCA 2

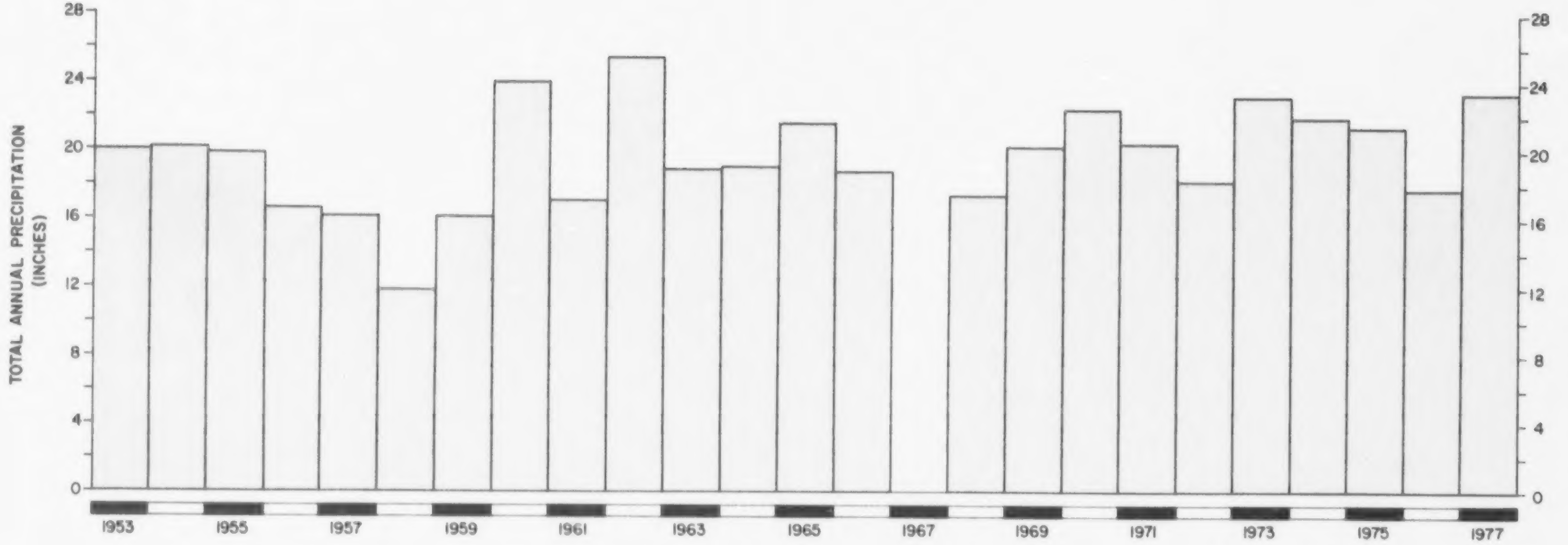


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FIG. 14 LONG-TERM ANNUAL PRECIPITATION AT ATHABASCA 2

Table 6. Mean daily global solar radiation (Langleys)

Month	Intensity
January	75 - 100
February	150 - 200
March	250 - 300
April	400 - 450
May	450 - 500
June	500 - 550
July	500 - 550
August	400 - 450
September	250 - 300
October	150 - 200
November	75 - 100
December	50 - 75

5.0 HYDROLOGY

5.1 SURFACE RUNOFF

Selection of a suitable water year is important for hydrological analyses. The ideal water year is that successive 12 month period that most consistently gives highest correlation between precipitation and streamflow (Likens et al. 1977). The water year March 16 to the following March 15 was established for all streams in the Baptiste watershed primarily for logistical convenience.

Flow regimes may vary because of seasonal and annual variations in climate, and because streams differ in size and in watershed land use. In years with average preciptation, some streams flow only in response to heavy, short-term rain events or runoff resulting from the spring melt. A pronounced peak in discharge occurs over a short period of time. These are known as ephemeral or intermittent streams and in the Baptiste watershed streams "B", "C", "H", "I", "J", "K" and "M" are classified as such. Perennial streams such as "A", "D", "E", "F", "L" and "N" are larger and more stable. Surface runoff in the latter extends over a much longer period of time and does not reveal such rapid, pronounced discharge peaks (Ward 1975).

In general, streamflow in the Baptiste watershed occurred on approximately 260 days each year and ranged from zero during dry periods to over 300 dam^3 day^{-1} during specific storm events (1 dam^3 = 1000 m^3). During spring runoff, the maximum daily discharge in the larger streams amounted to approximately 100 dam^3 day^{-1};

heavy discharge in early April was characteristic of all influent streams. Occasional summer or autumn storms also caused very high discharges in the perennial stream. They maintained a flow during each summer of observation, and daily discharges were generally less than 25 dam^3 day^{-1}. In winter, streamflow was diminished as streams froze and precipitation accumulated as snowpack. The outlet stream froze completely during the 1976-77 winter, but remained open during the winters of 1977-78 and 1978-79 as a result of higher lake levels.

The total annual discharge (m^3 yr^{-1}), areal hydraulic yields (m^3 ha^{-1} yr^{-1}), and mean annual hydraulic yields for the Baptiste streams from 1976 to 1978 are presented in Table 7. There was an approximate three-fold increase in total annual discharge from streams to the lake between 1976 and 1978. A consistent year to year increase in yields was not apparent for ephemeral streams but was for the larger, more stable streams. Ninety-one, eighty-eight and ninety-five percent of the total annual discharge to the lake during 1976, 1977, and 1978 respectively was contributed by streams "E", "F", and "L" (streams of large forested basins). Stream "G" (groundwater discharge) showed little annual variation in discharge.

Areas immediately adjacent to a lakeshore and which drain directly into the lake without entering a stream basin are known as diffuse runoff areas. Runoff from these areas is derived mainly from overland or seepage flow and may be estimated from adjacent stream basins assuming that similar near-surface conditions exist in both terrestrial areas (Uttormark et al. 1974).

Table 7. Total annual stream discharges (m^3) and areal yields ($m^3\ ha^{-1}\ yr^{-1}$).

Stream	1976-77	Areal Yield	1977-78	Areal Yield	1978-79	Areal Yield
A	119,688	134	392,853	440	488,265	547
B	10,500	125	70,851	845	29,619	353
C	75,838	267	171,880	606	134,737	475
D	327,467	447	489,538	668	672,571	917
E	5,736,561	560	4,016,481	392	14,766,434	1441
F	2,593,827	459	3,808,986	673	6,181,863	1093
G	50,596	-	48,776	-	48,776	-
H	69,059	800	13,187	153	29,343	340
I	166,321	465	199,673	559	143,782	402
J	41,229	370	91,650	823	44,314	398
K	142,234	460	60,038	194	49,655	161
L	1,359,991	153	3,347,545	376	8,135,495	914
M	12,698	180	33,990	482	30,916	439
N	190,944	271	267,874	381	257,573	366
O	6,088,640	-	14,937,008	-	24,321,164	-
Total A-M	10,706,009	386	12,745,488	460	30,755,770	623

The estimated annual discharge from areas of diffuse runoff, calculated from stream basin areal yields are given in Table 8. Total monthly discharge data (all inlets combined) for 1976 to 1978 are presented in Table 9.

In 1976, approximately 60 percent of the total annual volume of discharge occurred during April and May. The same percent volume in 1977 was split more evenly between April, May, and June, after which input decreased considerably until winter, when total input was negligible. The 1978-79 water year was more unusual, in that approximately 60 percent of the discharge occurred during September and October as a consequence of prolonged and heavy storm events. Actual volumes in 1977 and 1978 during April, May, June and July were fairly similar although the percentage distribution of annual discharge would indicate otherwise.

The distribution of total surface water inputs to Baptiste Lake for the three study years is presented in Table 10. This table illustrates the annual variation in runoff to the lake and also discharge through the outlet stream. Total input to the lake varied by a factor of three over the period of study and discharge through the outlet varied by a factor of four.

The south basin of Baptiste Lake consistently received more discharge throughout the study period than did the north basin. Nine streams and 7.5 diffuse runoff areas draining 63% of the watershed contribute to the south basin of Baptiste Lake, whereas only 4 streams and 5.5 diffuse runoff areas draining 37% of the watershed areas contribute to the north.

Table 8. Estimated annual discharge ($m^3 \, yr^{-1}$) from diffuse runoff areas.

	1976-77	1977-78	1978-79
1	3,006	14,897	10,435
2	15,738	58,158	33,201
3	8,887	13,285	18,252
4	6,924	10,168	16,502
5	50,492	60,617	43,650
6	12,964	2,475	5,508
7	2,413	6,461	5,876
8	7,743	25,416	31,589
9	104,765	156,614	215,170
10	16,232	23,837	38,686
11	4,264	6,262	10,163
12	141,801	379,581	345,247
13	469,423	328,664	1,208,330
Total	884,652	1,086,435	1,982,609

Table 9. Monthly discharge (m^3) into Baptiste Lake (1976-1979)

Month	1976-77	%	1977-78	%	1978-79	%
Mar	-	-	400,013	3.6	349,302	1.1
Apr	5,125,073	46.9	1,906,898	17.1	2,353,675	7.4
May	1,842,370	16.9	2,601,053	23.3	2,405,398	7.6
Jun	517,957	4.7	2,543,244	22.7	2,260,839	7.1
Jul	439,135	4.0	1,467,883	13.1	1,846,305	5.8
Aug	1,442,995	13.2	579,680	5.2	2,499,481	7.9
Sep	713,131	6.5	620,094	5.5	11,625,892	36.7
Oct	495,000	4.5	821,537	7.3	7,215,233	22.8
Nov	329,494	3.0	226,958	2.0	1,118,006	3.5
Dec	4,031	0.04	7,218	0.06	4,020	0.01
Jan	4,170	0.04	4,020	0.04	4,020	0.01
Feb	3,753	0.03	3,618	0.03	3,618	0.01
Total	10,917,109	100.00	11,182,216	100.00	31,685,789	100.00

Table 10. Percent distribution of hydraulic loadings (m^3 yr^{-1}) through Baptiste Lake.

Year	Total Input	South Basin	%	North Basin	%	Outflow
1976-77	11,550,661	9,528,179	82.5	2,002,482	17.5	6,088,640
1977-78	13,831,883	9,572,855	69.2	4,259,028	30.8	14,937,008
1978-79	32,738,379	23,344,145	71.3	9,394,234	28.7	24,321,164

The seasonal fluctuations in discharge for stream "A", which drains a typical agricultural basin, and stream "E", which drains a typical forested basin, are illustrated in the stream nutrient loading section (Figure 57). Daily discharges for stream "A" and "E" ranged from 0 to 23 $dam^3\ day^{-1}$ and 0 to 296 $dam^3\ day^{-1}$ respectively from March, 1976 to March 1979. Both hydrographs were characterized by high discharge from March to the end of April and followed by a decline until May. Peak spring flows occurred during the first week of April in stream "A" and somewhat later (mid-April) in stream "E". Occasional summer and autumn storms resulted in higher discharges which were generally sustained only for a short period. At times, high lake levels (resulting from high inflow to the lake due to storm events) made it impossible to measure discharge in stream "A" due to backflow. No discharge was recorded from late December to the beginning of March in any tributary stream.

Seasonal fluctuations in discharge through the outlet from 1976 to 1979 ranged from 0 $dam^3\ day^{-1}$ during the 1976-77 winter to as high as 302 $dam^3\ day^{-1}$ in September, 1978. Peak flows during spring runoff usually occurred around mid to late April somewhat near spring ice breakup, however, the fluctuations were more gradual than in headwater streams. The outlet discharge is controlled primarily by lake elevation during the open water season and this was evident during periods of high runoff to the lake; a lag period of several days/weeks was often observed.

The stage discharge curve is utilized to produce a continuous record of stream discharge when a continuous record of stream elevation is available. Proper placement of gauges and recording devices in streams is critical since backwash from the lake and/or obstructions may affect the shape of the discharge-stage relationship.

In this study, high lake levels interfered with the establishment of good annual discharge rating curves for streams "D" and "E", beaver dams and macrophyte growth interfered in streams "D", "F", "L", and the outlet, and unreliable (too low discharge) data were available for streams "D" and "N". When necessary, instantaneous discharge values were estimated using sections of the rating curves that were developed.

5.2 LONG-TERM SURFACE RUNOFF

Estimating the runoff volume for stream "E" from miscellaneous discharge measurements resulted in an observed yield of 37.58 dam^3 km^{-2} for the period of March to October, 1977. This compared favourably with the yield of 33.77 dam^3 km^{-2} observed at Dapp Creek during the same period. Since the stream "E" estimate is based on limited data, the deviation of 3.81 dam^3 km^{-2} is not considered significant enough to warrant further adjustments to the following derived estimates of long-term runoff volume. This observation also helps to validate the annual discharge rates and annual nutrient loads that have been calculated for the various tributaries based on miscellaneous measurements.

Table 11 presents the long-term annual yields (Nov. 1 - Oct. 31) for Dapp Creek from 1967 to 1978 and the runoff volume calculated for the Baptiste Lake watershed for those same years.

Table 11. Calculated runoff into Baptiste Lake.

Year	Dapp Creek Annual Yield ($dam^3\ km^{-2}$)	Calculated Baptiste Inflow Runoff Volume (dam^3)
1967	33.03	9,810
1968	14.74	4,380
1969	41.31	12,270
1970	45.25	13,440
1971	88.42	26,260
1972	62.29	18,500
1973	22.42	6,600
1974	100.47	29,840
1975	29.02	8,620
1976	22.42	6,600
1977	35.15	10,440
1978	55.68	17,200
Average for period	45.85	13,670

One point to note is that during the study years of 1976 and 1977 runoff at Dapp Creek was below the long-term average even though 1977 was a higher than average precipitation year. Runoff was above the long-term average during 1978.

5.3 EVAPORATION

Evaporation data for Edmonton in 1976 and 1977 were 637 mm and 587 mm respectively. Baptiste Lake evaporation for these years is, therefore, estimated to be 561 mm and 517 mm. The SNBB estimates for average long-term lake evaporation in the Baptiste Lake area is 610 mm annually, which would remove approximately 5500 dam^3 yr^{-1} from the lake surface.

5.4 PRECIPITATION

The long-term annual precipitation inputs to Baptiste Lake based upon the water year November 1 to October 31 are presented in Table 12.

5.5 GROUNDWATER

Two deep test-hole drilling programs were conducted during 1977 and 1978 to investigate the lithographic and stratigraphic characteristics of the area, and to determine if the buried pre-glacial bedrock valleys reported by Borneuf (1972) were actually present. The existence of the bedrock channels was confirmed on either side of the lake, but there was no evidence to show that they actually interface with the lake basin, and in fact, analysis of the field data indicated that they may pass below the base of the lake.

Table 12. Calculated annual precipitation input to Baptiste Lake

Year	Precipitation input to Baptiste Lake (dam^3)
1967	3,140
1968	4,300
1969	4,700
1970	5,140
1971	4,670
1972	4,430
1973	4,860
1974	5,830
1975	4,780
1976	4,170
1977	5,630
1978	5,040
Average for period	4,720

Three observation wells were completed in the sand and gravel unit situated at the base of each valley (Figure 9). These wells were used to monitor groundwater pressure and chemistry and to determine the hydraulic properties of the aquifer. Flows were calculated using Darcy's Law (Davis and Dewiest 1966).

Minimum and maximum values of the hydraulic gradient (I) and the cross-sectional area (A) for each bedrock channel were determined; these figures were then used with the hydraulic conductivity (K) to provide a range of flows (Table 13). Therefore, the range of flows expected from the bedrock channels using this technique is 0.368 to 15.032 $dam^3\ yr^{-1}$.

During early May 1978, 23 water table wells were installed to monitor the fluctuations and chemistry of the shallow water table zone. The exact depth of the water table was difficult to determine in most of the wells completed in the clay-rich till, although the zone of saturation was often found less than 10 m below the ground surface. Estimates of potential groundwater flows through the various surficial deposits are presented in Table 14. These estimates assume that each surficial unit at the point of its discharge into the lake is 6 m thick; in addition, two individual units of each type of deposit were present with the exception of the lacustrine till. A range of values for each parameter in the Darcy equation was not available for this component; therefore these data represent only a crude estimate, with actual flows probably falling within an order of magnitude above or below these figures. Thus, the expected flow using this approach may be in the range of 63 to 6300 $dam^3\ yr^{-1}$.

Table 13. Estimated groundwater inflows via bedrock channels.

	K ($m\ day^{-1}$)	A (m^2)	I	Q ($m^3\ day^{-1}$)	Annual Flow dam^3
N.W. Channel:					
min:	.015	8,000	.0024	.29	.105
max:	.015	12,000	.0038	.68	.249
S.W. Channel:					
min:	.108	3,200	.0002	.72	.263
max:	.108	24,000	.016	40.50	14.783

Table 14. Estimated groundwater inflows from surficial deposits.

	K ($m\ day^{-1}$)	I	A (m^2)	Q ($m^3\ day^{-1}$)	Annual Flow dam^3
aeolian	8.6	.021	7,800	1,409.0	514.3
deposits	8.6	.020	1,800	310.0	113.2
till	.0086	.021	12,100	2.2	.803
	.0086	.020	28,800	5.0	1.825
till (reworked	.0009	.023	46,000	0.1	.037
by water)	.0009	.023	4,500	0.09	.033
lacustrine-	.0051	.020	6,600	0.60	.219
alluvial	.0051	.023	9,300	1.07	.391
till-	.0086	.020	14,900	2.6	.949
alluvial	.0086	.020	13,000	2.2	.803
lacustrine till	.0008	.023	14,800	0.3	.110
total flow					632.670

The main groundwater recharge area for the watershed is the upland areas to the south and west of the lake. While most of the precipitation is discharged directly to the lake as surface runoff, some is available for recharge to the groundwater system. Water entering the shallow glacial deposits is primarily discharged to streams in the watershed, or directly to the lake. Some of this groundwater percolates downward into the deeper groundwater system, consisting of the deeply buried till, which discharges to Baptiste Lake on a long-term basis. While some groundwater may discharge from the buried valley to this lake and vice versa, it is proposed that much of this groundwater continues to flow eastward beneath the lake. Groundwater outflow from the lake also occurs through the glacial deposits on the east side of the lake (Crowe and Schwartz 1980).

5.6 GROUNDWATER MODELLING STUDY

A complete description of the operation of the groundwater model can be found elsewhere (Crowe 1979). The following discussion is a synopsis of that work.

Most hydrogeological studies of lake-watershed systems have relied on conventional water-balance methods (Laycock 1968, 1971, 1973; Nursall et al. 1971). Fritz and Krouse (1973), using dissolved sodium and natural stable isotopes of oxygen and hydrogen (oxygen-18 and deuterium), evaluated the pattern of groundwater inflow to a prairie lake and characterized the evaporitic regime. Schwartz and Gallup (1978) demonstrated how major ions can be utilized to determine the relative importance of groundwater recharge and discharge to prairie lakes. These studies did not take into account the time-varying changes that occur in a system, such as routing water and major ions through components of a prairie lake-watershed system on a monthly or daily basis, but merely predicted an average lake level or chemistry from the contributing components independent of the time when the data were collected.

A logical extension of the above chemical balance technique lies in the routing of major ions through a lake-watershed system over a period of time. This is in fact the basis for the lumped response model developed by Crowe (1979) which simulates the hydrological cycle by routing water plus ions through storage units representing different hydrostratigraphic components of the watershed.

The groundwater system in the Baptiste Lake watershed was represented by eight groundwater storage elements; five shallow and three deep. They were: elements 1 and 2, which represented the alluvium-moraine deposits with and without an overlying unsaturated zone, elements 3 and 4 which corresponded to the shallow moraine units, again with and without an overlying unsaturated zone; element 5, which represented the surficial deposit of sand; element 6, the deeply buried moraine; and two elements, 7 and 8, which represented the sand and gravel in the north and south buried channels, respectively.

Rivers were not as fully constructed by the model as they would be in the real hydrologic environment. Only the groundwater discharge component of stream flow was computed with the model. Groundwater discharged to a river was a separate parameter because one of the objectives of the model was to determine how much water entered the lake directly and how much entered the lakes indirectly via rivers. It is assumed that all surface runoff and impervious flow would reach the lake whether or not it entered a river, and hence, that these two components were routed directly to the lake. River flow can be estimated by summing surface runoff from impervious areas and groundwater discharge to rivers.

The pathways of outflow from the lake were via the surface and subsurface. Surface water outflow was calculated using only the volume of lake water above the surface water outlet elevation. A capacity-discharge curve function for the lake should normally be provided by the model user. Outflow via groundwater was simply removed from the lake, in a similar fashion to surface water outflow, rather than by routing it through groundwater storage elements.

The hydrological variables which the model simulated were: lake stage, salinity of the lake, outflow from the lake via groundwater and surface water, recharge to the groundwater zone, unsaturated zone moisture, thickness of ice during the winter, surface runoff, and groundwater discharge to the lake. Where possible, the parameters required for simulation were taken from real data. However, some could only be estimated by trial and error (Crowe 1979; Crowe and Schwartz 1981(a)).

5.7 RESULTS OF GROUNDWATER MODELLING STUDY

A satisfactory simulation of lake surface levels and lake salinity fluctuations at Baptiste Lake was achieved with the model. The predicted and measured lake levels had a mean absolute difference of less than 0.1 meter, and the mean difference between simulated and actual lake salinity was approximately 7 mg L^{-1}. Although the actual and calculated values were not always the same, the patterns of fluctation followed the same trends. The discrepancies between the two sets of results were probably caused by: i) the simplified form of the model, ii) data accuracy, and, iii) inaccuracy of the hydrologic function.

Table 15 lists the percentages of various inflows and outflows to Baptiste Lake for the period of 1972 to 1978 as calculated with the model. During this time, the groundwater contribution to Baptiste Lake averaged 13%.

To further verify those results, the watershed was modelled assuming no groundwater discharge to the lake or lake discharge via groundwater. All water was assumed to enter the lake as either surface runoff or as precipitation falling directly on the lake surface. The resulting lake level fluctuations were similar to the actual values. However, a closer fit was obtained when the groundwater component of the watershed budget was included as part of the discharge to the lake. Furthermore, lake salinity without groundwater input was less than half of the salinity produced when groundwater discharge was included in the simulation. Also, the range of salinity fluctuations that existed at Baptiste Lake could not be predicted without the simulation of groundwater discharge. This result presented a striking illustration of the importance of the groundwater contribution to Baptiste Lake (Crowe 1979; Crowe and Schwartz 1981(b)).

Table 15. Proportions of the hydrologic balance at Baptiste Lake.

	% Inflow			% Outflow		
	Ground Water	Surface Water	Precip.	Ground Water	Surface Water	Evap.
1972	22	68	10	13	72	15
1973	13	68	19	14	70	16
1974	18	74	8	8	83	9
1975	9	74	17	14	69	17
1976	7	76	17	18	60	22
1977	9	75	16	13	72	15
1978	12	71	17	16	68	16

Table 16. Total annual water inputs to Baptiste Lake (m^3 yr^{-1})

	1976	1977	1978	Long-Term
Streams	10,706,009	12,745,448	30,755,770	13,670,000
Diffuse Runoff	884,652	1,086,435	1,982,609	
Precipitation	4,155,337	5,445,727	5,442,449	4,720,000
Groundwater	1,185,183	1,906,557	5,206,447	2,993,720
Total	16,931,181	21,184,187	43,387,305	21,383,720

5.8 HYDRAULIC RESIDENCE TIME

The volume of the lake, based on the existing area-capacity curve, was estimated to be 85,500 dam^3 at an elevation of 578.35 m GSC. This curve could only furnish an estimate accurate to ± 10%; therefore, the expected lake volume ranges between 76,950 dam^3 and 94,050 dam^3.

The hydraulic residence time of a water body is the ratio of its volume to the annual inflow, and represents its filling time. Surface and subsurface runoff as well as precipitation account for the inflow parameters, and the total inflow volumes for the study period and for the long-term (1967-78; November 1 - October 31) are present in Table 16.

Therefore the long-term residence time $= \frac{\text{lake volume}}{\text{inflow volume}}$

$$= \frac{85{,}500\ dam^3}{21{,}380\ dam^3\ yr^{-1}}$$

$$= 4.00 \text{ years}$$

If the ± 10% lake volume range is used, the replacement time would range from 3.60 to 4.40 years.

5.9 HYDROLOGY OF GHOST AND ISLAND LAKES

The principal hydrologic and morphometric features of Ghost and Island Lakes are summarized in Table 17. The surface runoff and precipitation inputs to these waterbodies have been derived using the same data incorporated in the calculated long-term water inputs to Baptiste Lake.

Table 17. Hydrology and morphometry of Ghost and Island Lakes.

	Island	Ghost
Watershed Area (ha) Excluding Lake	3806	2201
Lake Surface Area (ha)	778	199
Lake Volume (dam^3)	29066	9006
Mean Depth (m)	3.74	4.52
Annual Precipitation Inputs (dam^3)	3990	1036
Annual Surface Runoff (dam^3)	2830	990
Total Annual Inflow (dam^3)	6720	2026
Hydraulic Residence Time (yr)	4.33	4.45
Areal Water Loading (m yr^{-1})	0.86	1.01

The watershed area presented in Table 17 for Island Lake excludes Ghost Lake. These lakes receive low areal water loadings, but because of their volumes have moderately long rates of water renewal. Subsurface water inputs are not accounted for in these calculations (Bothe 1978).

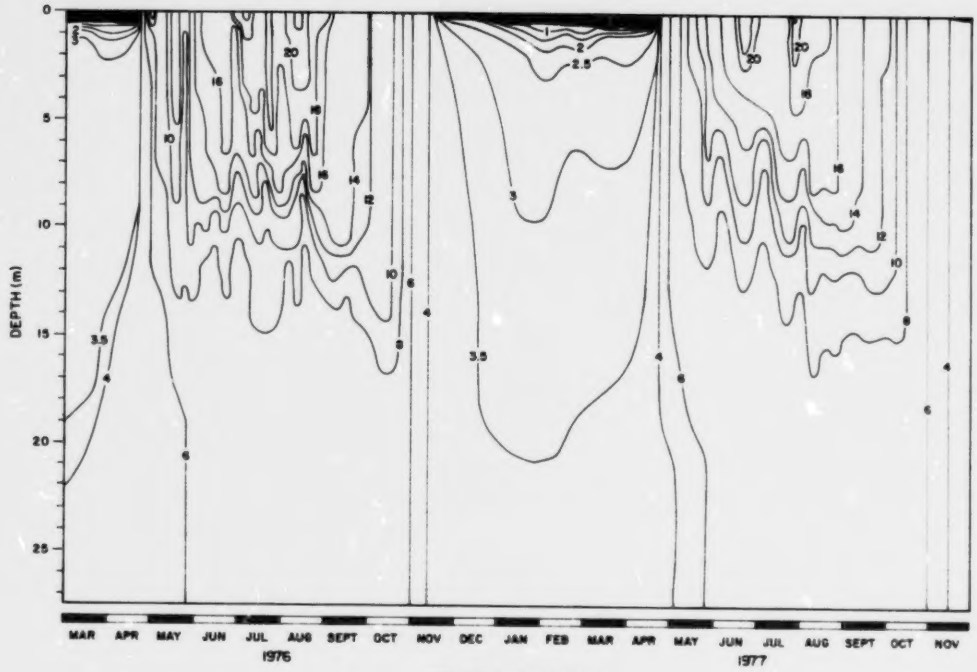
6.0 PHYSICAL-CHEMICAL CHARACTERISTICS

6.1 TEMPERATURE, OXYGEN AND CONDUCTIVITY

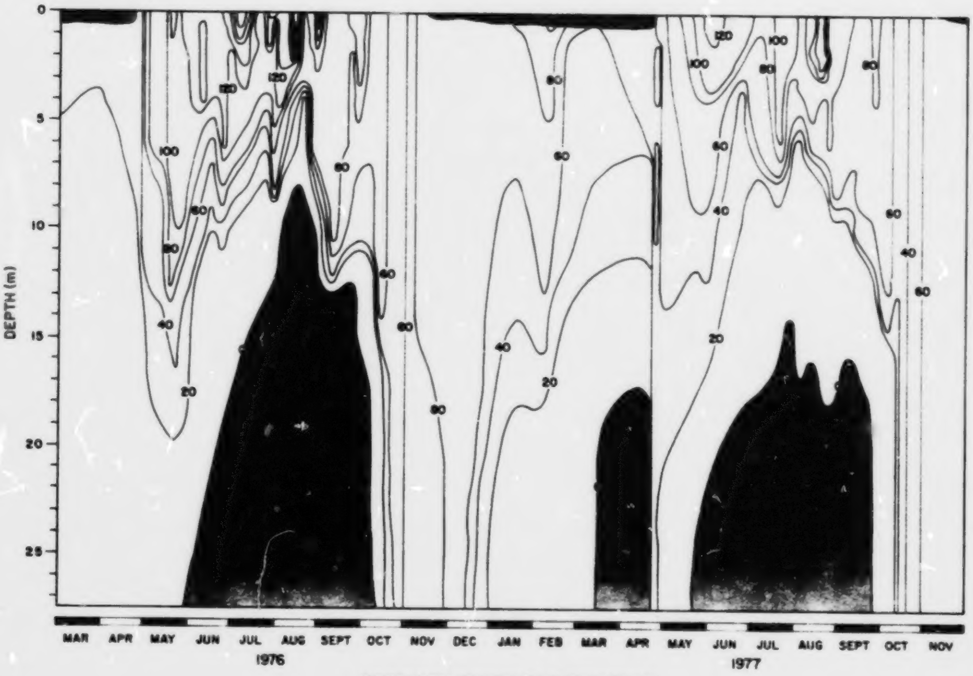
Vertical profiles of temperature, dissolved oxygen, and conductivity are summarized in Figures 15 to 17. These time-depth diagrams represent a concise and simple method of presenting the changes in these parameters at each sampling site. The sites were assumed to be representative of the horizontal distribution of these parameters throughout the water column in each basin.

Unlike most temperate, dimictic lakes, Baptiste Lake appears to exhibit partial meromixis. That is, it does not undergo two periods of complete vertical mixing each year. Observations indicate that the deeper south basin did not mix thoroughly during the spring of 1976, while the shallow north basin exhibited a delay in mixing until late May of that year. The 1977 spring turnover was complete in both basins.

The physical and chemical data support this interpretation. In the north basin, temperature profiles under the ice cover in late March of 1976 indicated that bottom temperatures near the sediments were elevated above the temperature of maximum density (4.0°C); however, this was not observed in the south basin. According to Wetzel (1975) this phenomenon can be the result of the release of heat stored in the sediments from the summer period, which one would anticipate to occur more readily in shallow basins. It can also occur from density currents generated by the the solar heating of



TEMPERATURE (°C)



DISSOLVED OXYGEN (% SATURATION)

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FIG. 15 TIME-DEPTH DISTRIBUTION OF ISOTHERMS (°C) AND ISOPLETHS OF DISSOLVED OXYGEN (% SATURATION) FOR BAPTISTE LAKE (SOUTH)

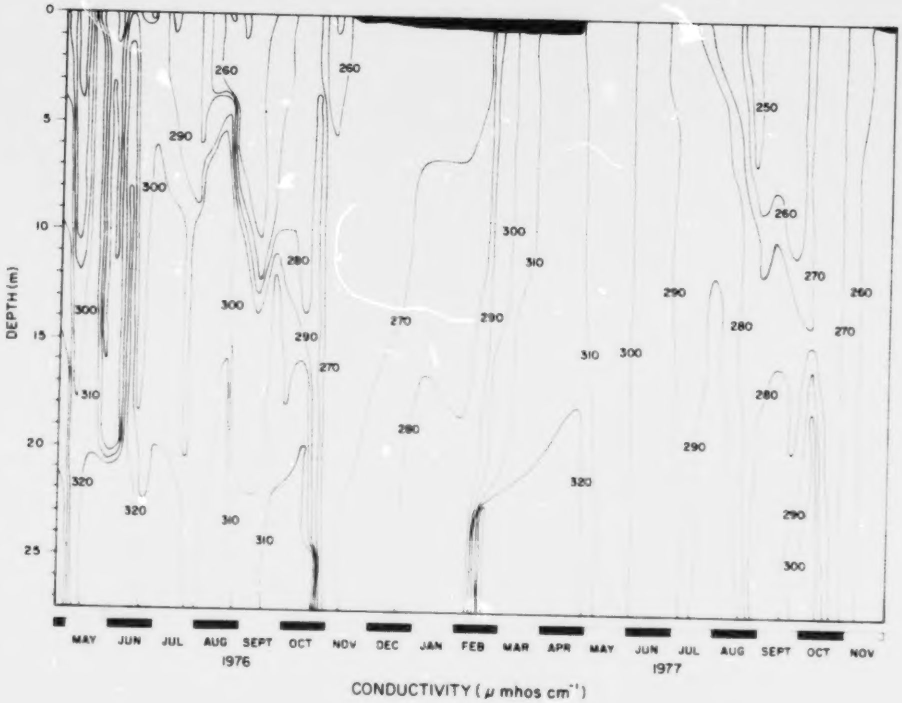

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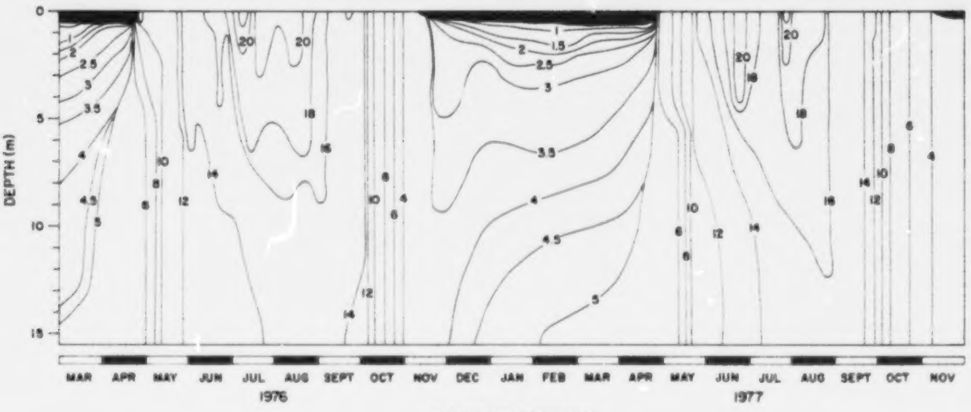
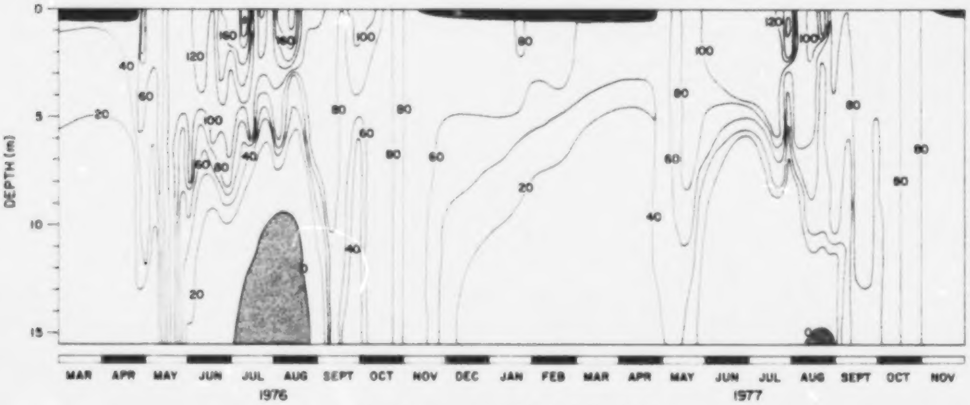
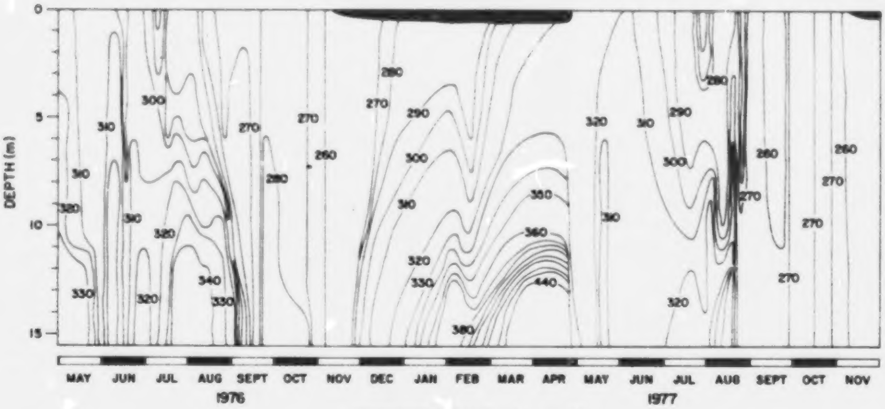
FIG. 16 TIME-DEPTH DISTRIBUTION OF ISOPLETHS OF SPECIFIC CONDUCTANCE (μmhos cm^{-1} at 25°C) FOR BAPTISTE LAKE (SOUTH)



TEMPERATURE (°C)



DISSOLVED OXYGEN (% SATURATION)



CONDUCTIVITY (µ mhos cm⁻¹)

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FIG. 17 TIME - DEPTH DISTRIBUTION OF ISOTHERMS (°C), ISOPLETHS OF DISSOLVED OXYGEN (% SATURATION), AND ISOPLETHS OF SPECIFIC CONDUCTANCE

water through the ice in shallow littoral areas, which flow toward the central portion of the basin over the sediments. Thermal-density instability is subsequently prevented by the diffusion of solutes from the sediments into these waters.

A further possible explanation for elevated temperatures at this location is groundwater seepage. The mean annual temperature of groundwater in this region is likely to be greater than 4°C (D. Prosser, Earth Sciences and Licencing Division, Alberta Environment, personal communication), and the work described previously (Crowe and Schwartz 1981) has provided some evidence for groundwater flow into the lake at this location. This point will be discussed further in the chemical section.

Chemical analysis of lake water sampled from the north basin in mid-March of 1976 indicated a pronounced vertical conductivity gradient ranging from 325 micromhos cm^{-1} just under the ice to 375 micromhos cm^{-1} near the sediments, which corresponded to a vertical gradient in total dissolved solids of 20 mg L^{-1}. An in situ conductivity profile taken on April 28, 1976, three days after ice-out, showed a gradient of approximately 25 micromhos cm^{-1} which corresponded to a solute gradient of 12 mg L^{-1}. Small changes in dissolved solids concentration can effect very significant changes in specific gravity relative to thermally induced specific gravity changes near 4.0°C; this "concentration stability" could then have provided a resistance to mixing (Wetzel 1975).

The oxygen profile recorded on April 28, 1976, also indicated a pronounced gradient, a condition which existed until late May. One would have expected fairly uniform oxygen conditions throughout the water column had mixing occurred normally at ice-out, as is observed in both basins during the period of fall mixing. Only an exceptionally high oxygen demand in the water column or from the sediments could otherwise account for this situation.

The temperature profile recorded on the same day indicated that the water column was sharply stratified near the surface due to the extremely warm weather that followed ice-out that year. Bottom temperatures were 5.0°C which would indicate that 4.0°C water had not been mixed down from the surface, or that mixing had occurred and the entire water column was warmed very quickly by the weather conditions of that week. However, the high surface temperatures suggest that stratification had been rapid and undisturbed by much wind activity.

By May 19, 1976 the water column in the north basin had become nearly isothermal at approximately 11.0°C, with oxygen and conductivity also being similar through all depths. The water column was less stable at this time, and it appears that the combination of stronger winds and stabilized air temperatures had permitted a brief period of circulation. The basin was weakly stratified during the summer, becoming isothermal by September 8, 1976.

In the south basin, the conductivity profile on April 28, 1976 indicated a 60 micromho cm^{-1} gradient between top and bottom corresponding to a measured solute difference of 21 mg L^{-1}. The temperature profile showed a rapid increase occurring near the surface with bottom temperatures at 3.8°C, slightly above the mid-March value of 3.5°C, indicating that the warmer surface waters may not have reached this depth. The most dramatic evidence for a lack of circulation is the oxygen profile which displayed a pronounced vertical gradient with less that 20% saturation below 12 m. This gradient remained intact for the entire summer. The south basin exhibited a pronounced thermal stratification extending from early June until fall turnover in mid-October. This was followed by approximately four weeks of isothermal conditions during which the water column circulated freely and the hypolimnetic waters were re-oxygenated.

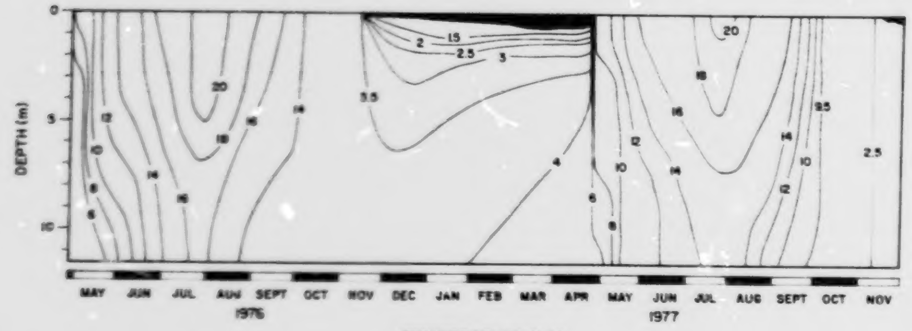
During the winter of 1976-77 the ice cover reached a maximum thickness of 60 cm, and once again the bottom waters of the shallow north basin exhibited temperatures up to 5.0°C near the sediment. This was accompanied by a significant gradient in specfic conductance (Figure 17). The potential was therefore available for the basin to again mix incompletely; however, strong spring winds mixed the water column normally. In 1977, summer stratification commenced in mid-June in both basins and the temperature cycle was fairly similar to the pattern of 1976, although temperatures were slightly lower. The north basin became isothermal by August 24 of that year.

The summer oxygen conditions in Baptiste Lake were characterized by an extreme vertical gradient in saturation. The hypolimnetic waters of the south basin underwent a rapid and prolonged oxygen depletion. The depletion extended until just prior to fall turnover in October, and the maximum vertical extent of this anoxic zone was between 8 m and the bottom during August, 1976.

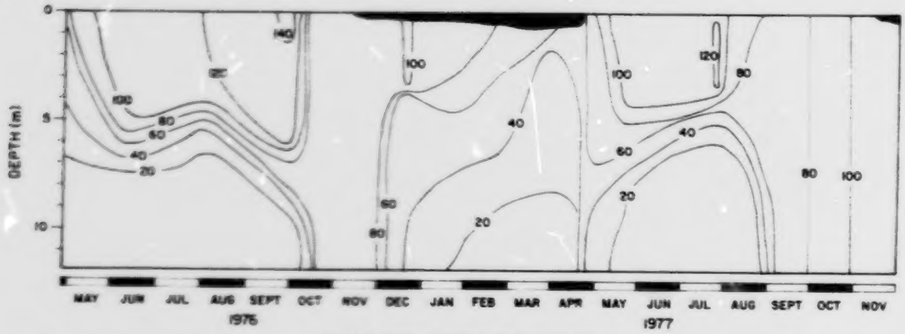
During the same year, the north basin underwent a shorter period of deoxygenation extending from early July to the end of August, and this zone extended from 10 m to the bottom. The hypolimnetic waters are generally in poor condition except for the short intervals during which vertical mixing occurs.

The surface waters, however, were super-saturated for a large portion of the summer; this is probably due to the high photosynthetic activity of the phytoplankton. During the isothermal conditions in the fall a slight depression in oxygen saturation was evident, possibly a result of the high oxygen demand hypolimnetic waters being distributed throughout the water column. Prior to freeze-up there was a slight increase again. A similar pattern was evident in the north basin. Oxygen depletion was also observed in the south basin during the winter of 1976-77, although the duration was short and only detectable below 18 m. The general condition of much of the water column under the ice was one of low saturation. Spring turnover in late April 1977 allowed a certain amount of re-oxygenation to occur; however, it is apparent that a high oxygen demand exists in the water column since depletion occurs early in the growth season. The general conditions in the north and south basins in 1977 compared to 1976 were less extreme, with less supersaturation and a shorter period of depletion.

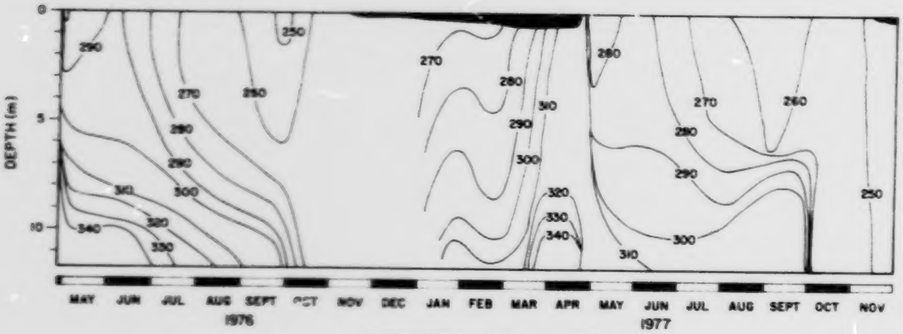
Ghost and Island Lakes (Figures 18 to 20) displayed mixing regimes similar to the basins of comparable depth in Baptiste Lake, although the monthly sampling interval allows for nothing more than a qualified description of trends.



TEMPERATURE (°C)



DISSOLVED OXYGEN (% SATURATION)



CONDUCTIVITY (μ mhos cm^{-1})

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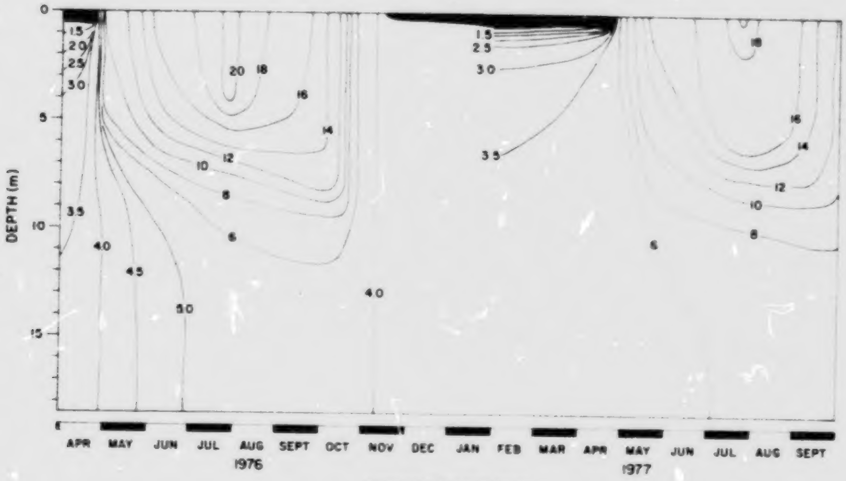
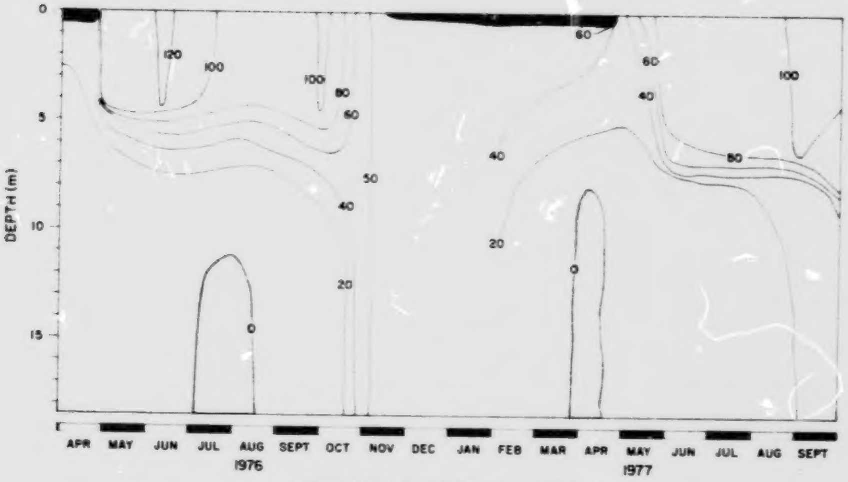
FIG. 18 TIME-DEPTH DISTRIBUTION OF ISOTHERMS (°C), ISOPLETHS OF DISSOLVED OXYGEN (% SATURATION), AND ISOPLETHS OF SPECIFIC CONDUCTANCE (μmhos cm^{-1} at 25°C) FOR GHOST LAKE



TEMPERATURE (°C)



DISSOLVED OXYGEN (% SATURATION)

FIG 19 TIME-DEPTH DISTRIBUTION OF ISOTHERMS (°C), AND ISOPLETHS OF DISSOLVED OXYGEN (% SATURATION) FOR ISLAND LAKE

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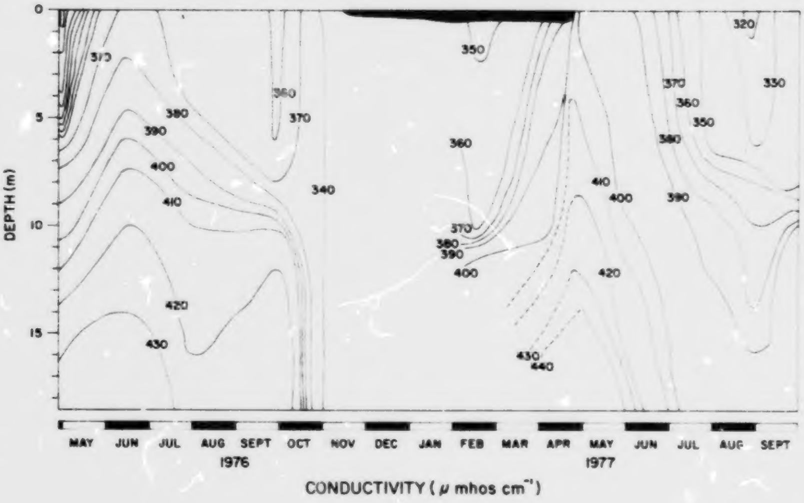

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FIG. 20 TIME-DEPTH DISTRIBUTION OF ISOPLETHS OF SPECIFIC CONDUCTANCE (μmhos cm⁻¹ at 25°C) FOR ISLAND LAKE

The sampling site on Ghost Lake was exposed to very little fetch, and our observations suggest that an incomplete vernal mixing occurred during 1976. However, no subsequent instability of the type observed in Baptiste Lake during late May of that year was detected in Ghost Lake. Mixing was apparently complete during the spring of 1977. A very weak thermal density stratification in summer is suggested by the temperature data, although vertical oxygen and conductivity gradients were detected.

Epilimnetic supersaturation of oxygen was observed during both growing seasons and hypolimnetic saturations were generally less than 20% during periods of stratification. Vertical conductivity gradients on the order of 50 micromhos cm^{-1} were also observed during these intervals.

The northern basin of Island Lake is also afforded protection from prevailing winds by the hills rising to the north and west. Our data suggest that this basin escaped spring turnover during both years of study; a pronounced thermal density stratification was in place and this partial meromixis may have been enhanced by a high conductivity gradient which appeared to develop in late winter of each year. Hypolimnetic oxygen saturations were at or very near zero during all periods of stratification.

Although information is missing on Island Lake for the late autumn of 1977, we assume that all basins underwent complete fall mixing due to the extended cooling period that is typical of the climate in this region.

6.2 MAJOR CATIONS

The seasonal fluctuations in major cations at surface (1.0 m) and bottom (1.0 m above sediments) sampling depths in each basin are illustrated in Figures 21 to 24. The pattern of cationic dominance throughout the water column of Baptiste Lake is $Ca^{++} > Mg^{++} > Na^{+} > K^{+}$, which is identical to that of Ghost Lake. This is characteristic of many other lakes in central Alberta (Mitchell 1979; Trew et al. 1981) and indeed as the principal composition of most temperate, open lake systems (Wetzel 1975). However, the pattern in Island Lake is $Mg^{++} > Ca^{++} > Na^{+} > K^{+}$. The surface, open water, mean concentrations for all three lakes, for all chemical parameters, are presented in Table 18.

The elevated concentrations of sodium in Baptiste Lake may be the result of groundwater intrusions from the buried bedrock valleys. According to Crowe and Schwartz (1981(b)), the groundwater in the two buried valley deposits and the overlying surficial deposits have very different chemical charateristics. The waters from the buried valleys have relatively high concentrations of Na^{+} and low concentrations of Ca^{++} and Mg^{++}. Waters from the other surficial materials are typically of a calcium-magnesium-bicarbonate type.

Magnesium, sodium, and potassium are relatively conservative both in their chemical reactivity under typical freshwater conditions and in their limited requirement for biotic activities (Wetzel 1975). This is generally the case in Baptiste Lake; pronounced seasonal fluctuations are absent except for the elevated sodium concentrations which were observed near the bottom

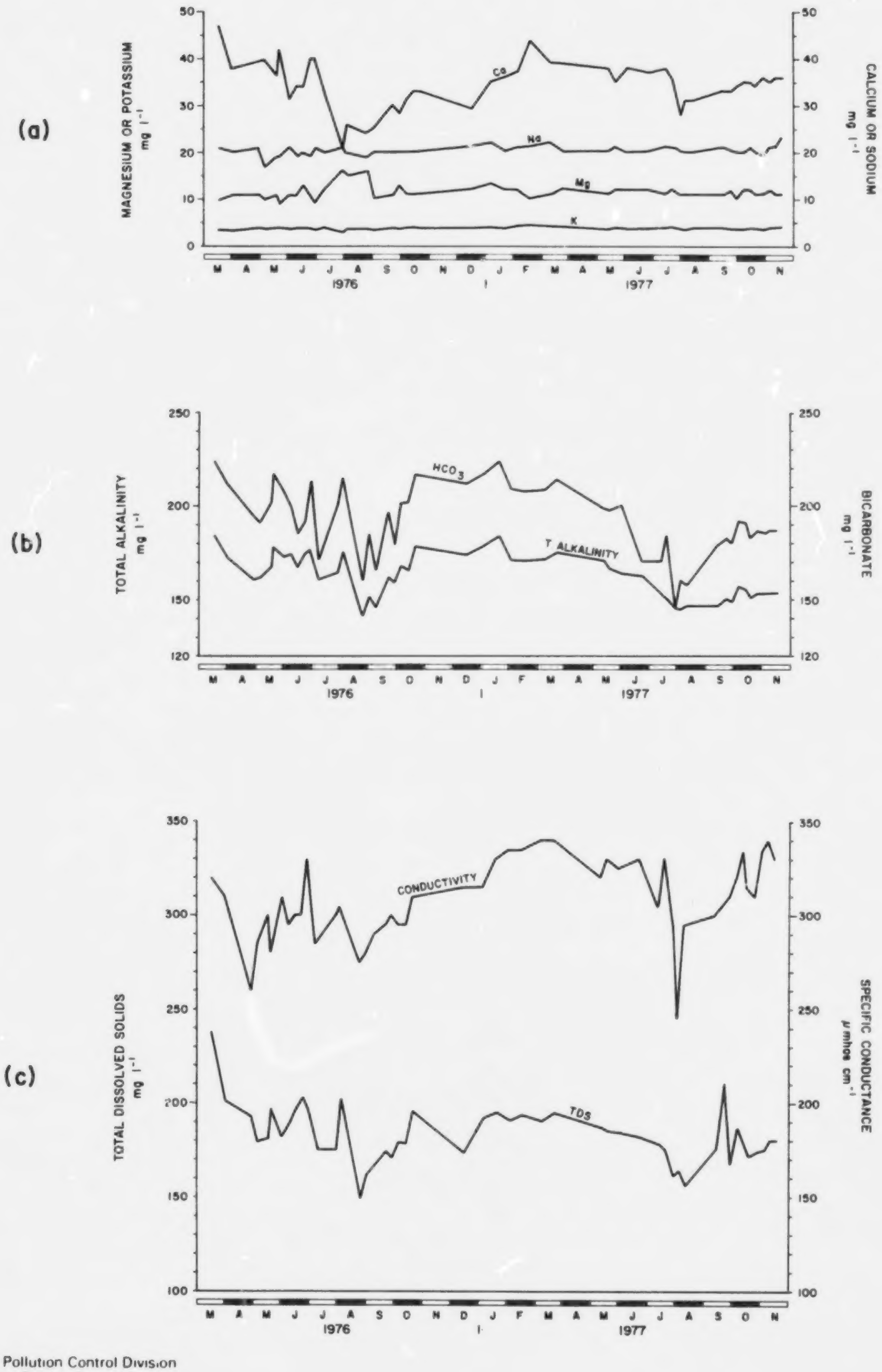

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FIG. 21 SEASONAL FLUCTUATIONS OF (a) MAJOR CATIONS, (b) TOTAL ALKALINITY AND BICARBONATE, AND (c) TOTAL DISSOLVED SOLIDS AND SPECIFIC CONDUCTANCE AT 0.5 m IN BAPTISTE LAKE (SOUTH)

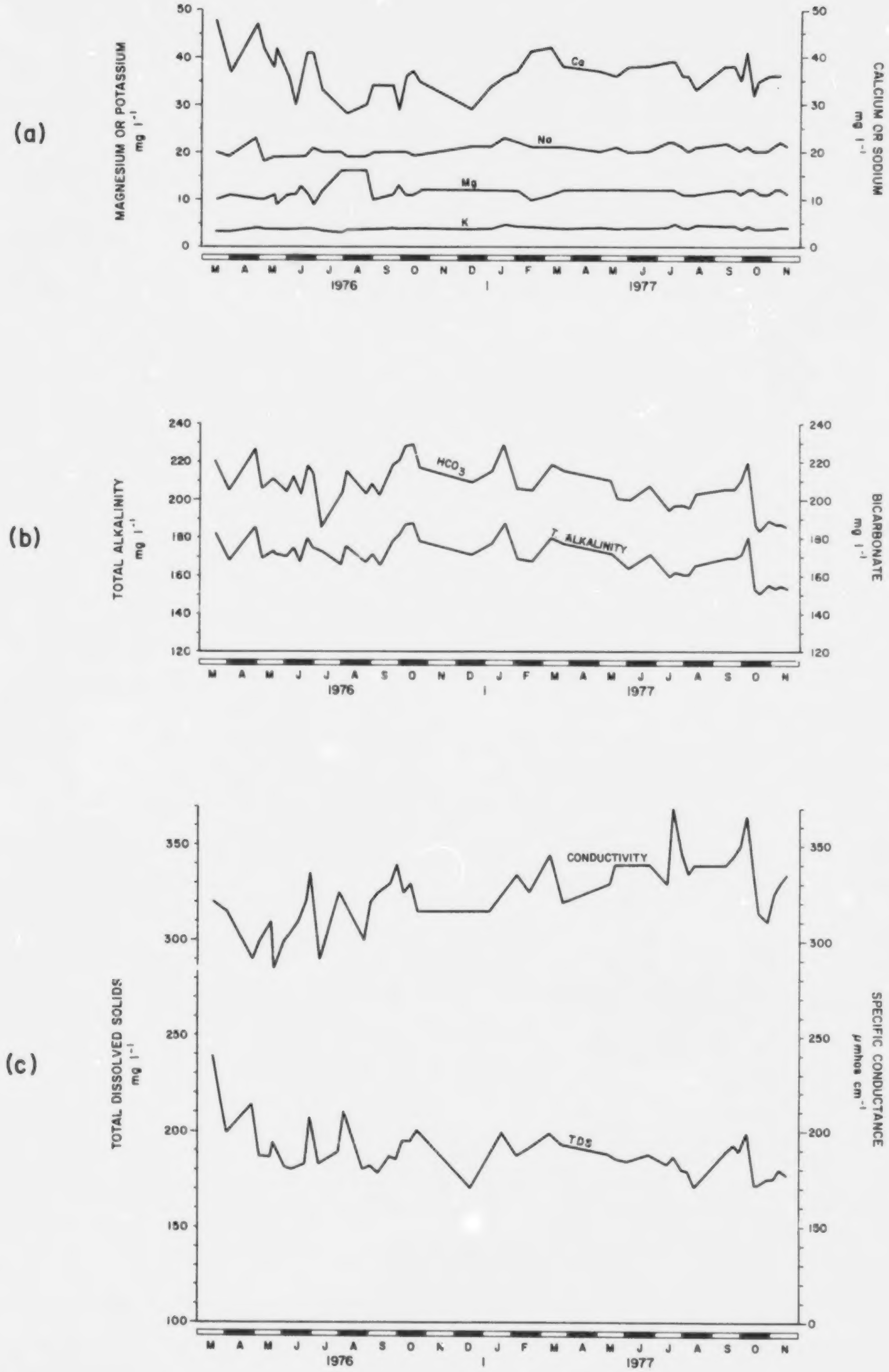

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FIG. 22 SEASONAL FLUCTUATIONS OF (a) MAJOR CATIONS, (b) TOTAL ALKALINITY AND BICARBONATE, AND (c) TOTAL DISSOLVED SOLIDS AND SPECIFIC CONDUCTANCE AT 22.0m IN BAPTISTE LAKE (SOUTH)

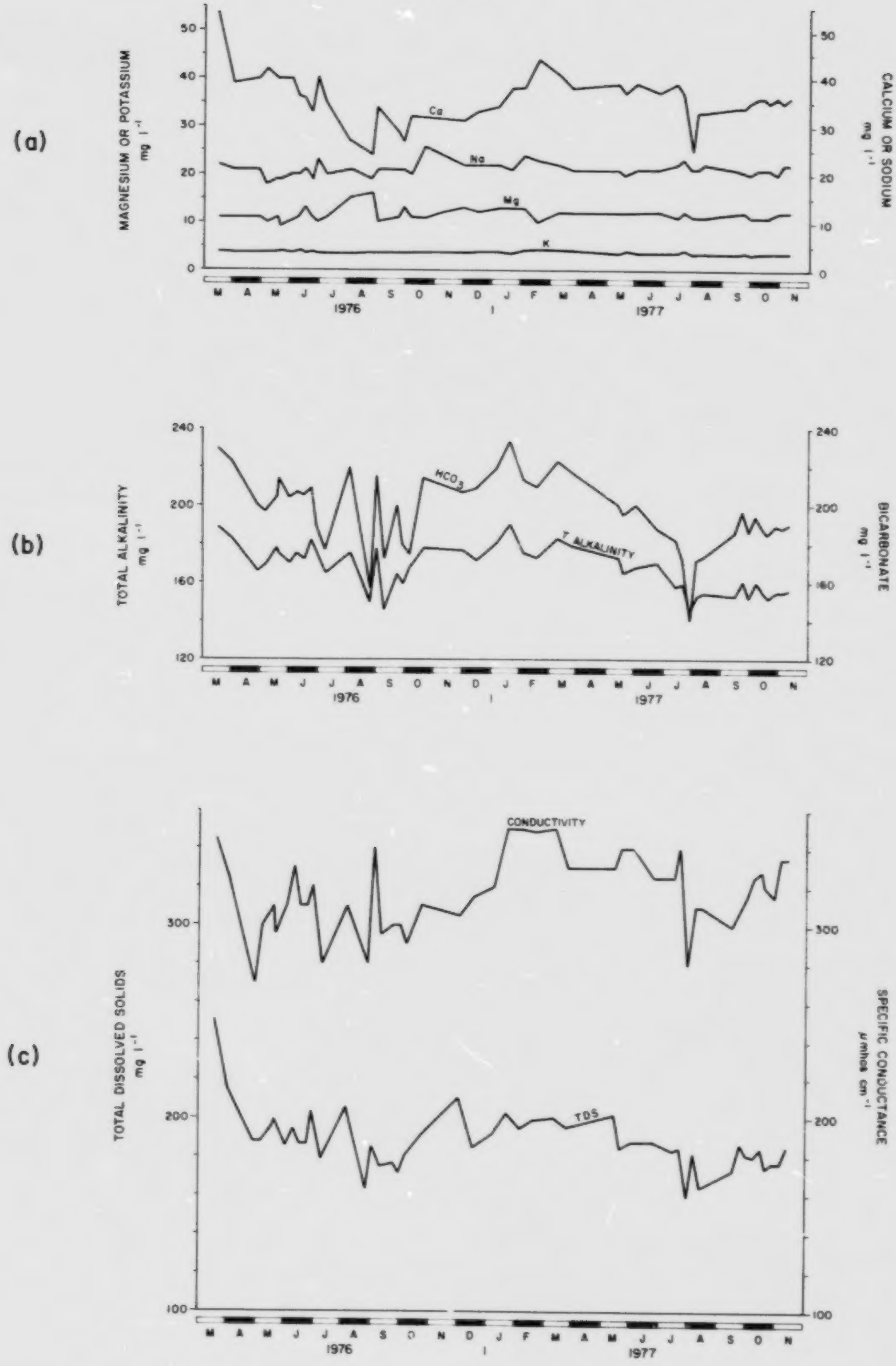

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FIG. 23 SEASONAL FLUCTUATIONS OF (a) MAJOR CATIONS, (b) TOTAL ALKALINITY AND BICARBONATE, AND (c) TOTAL DISSOLVED SOLIDS AND SPECIFIC CONDUCTANCE AT 0.5 m IN BAPTISTE LAKE (NORTH)

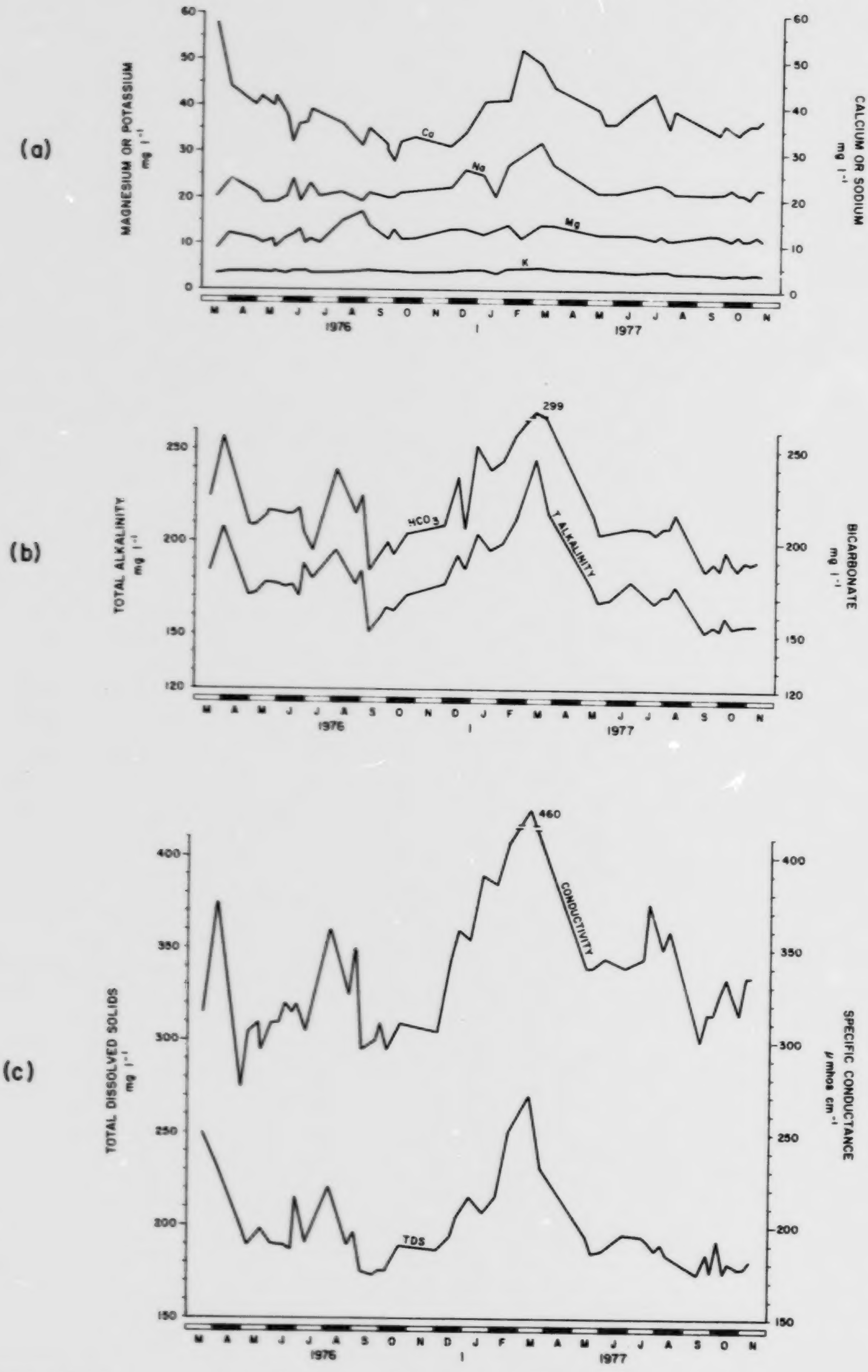

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FIG. 24 SEASONAL FLUCTUATIONS OF (a) MAJOR CATIONS, (b) TOTAL ALKALINITY AND BICARBONATE, AND (c) TOTAL DISSOLVED SOLIDS AND SPECIFIC CONDUCTANCE AT 12.0 m IN BAPTISTE LAKE (NORTH)

Table 18. Chemical Summary - open water means, 1977

Parameter	Baptiste L.	Ghost L.	Island L.
Conductance (μmhos cm^{-1})	320	300	370
Calcium (mg L^{-1})	35	38	25
Magnesium (mg L^{-1})	12	13	20
Sodium (mg L^{-1})	22	11	27
Potassium (mg L^{-1})	3.8	5.0	7.7
Iron (mg L^{-1})	0.07	0.05*	0.05*
Silica (mg L^{-1})	0.9	0.6	1.5
Chloride (mg L^{-1})	2.	2.	2.
Sulphate (mg L^{-1})	11.	10.*	10.*
Bicarbonate (mg L^{-1})	183.	202.	224.
Alkalinity (mg L^{-1})	156.	170.	186.
TDS (mg L^{-1})	180.	182.	206.
SRP (mg L^{-1})	0.020	0.008	0.010
Total Phosphorus (mg L^{-1})	0.070	0.05*	0.05*
TKN (mg L^{-1})	1.90	1.52	1.33
NH_3-N (mg L^{-1})	0.16		
NO_3-N (mg L^{-1})	0.042	0.026	0.017
TON (mg L^{-1})	1.74		

* indicates concentration less than

of the northern basin during February and March of 1977 (Figure 24).

By comparison, calcium is more reactive and can exhibit marked seasonal fluctuations. Calcium compounds are stable in water in the presence of carbon dioxide. The direct removal of CO_2 during photosynthesis can result in decalcification of the euphotic zone (Golterman 1975), and this appears to be the trend during July and August in Baptiste Lake when carbon fixation rates are at their maximum.

A trend towards high calcium concentrations is evident during winter of 1977 in the profundal waters of the north basin, coinciding with the observations on sodium at the same location. Reductions in calcium were observed in the profundal waters of both basins during the summer months of 1976 although the reasons for this are unclear.

6.3 BICARBONATE/TOTAL ALKALINITY

The total inorganic carbon pool was always dominated by bicarbonate in Baptiste Lake, even though pH values during the growth season were frequently above 8.3. Carbonate concentrations appeared to remain low, although one concedes that small changes in pH during sample storage could signficantly affect this parameter. In any event, bicarbonate may also be utilized directly during photosynthesis and as a result displays significant temporal oscillations (Figures 21 to 24). The surface observations in both basins during 1977 indicate a decrease from winter concentrations of aproximately 220 mg L^{-1} to late summer levels of 160-170 mg L^{-1}.

Following the trend of calcium and sodium, high concentrations of bicarbonate were observed during winter of 1977 near the bottom of the north basin. A temporary increase in bicarbonate during July-August of 1976 was also detected at this site, and corresponded to the only period of stratification that developed in the north basin during open water season of either year; no such fluctuations were observed in the south basin during summer or winter.

Bicarbonate was the dominant anion with both chloride and sulphate being near their lower detection limits on most occasions (1 mg L^{-1} and 10 mg L^{-1} respectively). Neither chloride or sulphate displayed perceptable changes through the year with the exception of slight increases in the profundal water of the north basin during winter, 1977.

Total alkalinity is a measure of a water's capacity to remove free hydrogen ions and is normally described as a function of bicarbonates, carbonates, and hydroxides. No data are presented with respect to the role of hydroxides in this system. The annual pattern in total alkalinity mirrors that of bicarbonate, and suggests that the latter may be the principal component of this lake's buffering capacity.

Baptiste Lake is well buffered, and this characteristic can be illustrated with a short anecdote regarding experiments carried out by Miller and Watkins (1946). Miller (1944) had observed from culture work that the coracidia of the tapeworm Triaenophorus crassus could be killed by lowering the pH to 5.0. Since this tapeworm eventually became an economic problem by infesting commercial fish stocks of whitefish and tullibee throughout western Canada, it was decided to undertake a whole-lake acidification experiment in an attempt to eradicate the parasite at this vulnerable coracidial stage and Baptiste Lake was selected. Twenty tons of sulphuric acid were added to the littoral zone in and adjacent to the shallow bays on the east side of the north basin. Immediately after addition the surface waters dropped below the range of their indicator (pH = 5.0), but within a half hour had returned to the normal range (pH = 8.0). The bottom sediments became very acidic, probably due to the fact that the acid sank with very little mixing. However, by the next day even the sediments had returned to their normal pH, suggesting that a strong buffering capacity is present in both water and sediment components.

6.4 SPECIFIC CONDUCTANCE/TDS

Specific conductance is a measure of the mobility of ions in solution, whereas total dissolved solids is a measure of the total amount of material in solution. The two parameters may or may not be related, depending upon the degree of dissociation of the various molecular constituents present.

The annual patterns for these two parameters are presented in Figures 21 to 24. Specific conductance in surface waters followed a general trend of maximum concentrations in winter and minimum concentrations in late summer, similar to bicarbonate. Ice formation has been shown in many cases to affect the former (Potter 1977), whereas photosynthetic influences on the carbon dioxide equilibria could affect the latter. As noted previously in the time depth diagram of specific conductance (Figure 27), a substantial late winter increase in this parameter was detected in the bottom waters of the north basin; no such phenomenon was observed in the south basin. Furthermore, a distinct increase was observed during the short summer stratification of July-August, 1976. The maximum conductance observed during these periods was 460 umhos cm^{-1}.

The fluctuations in total dissolved solids were generally similar to those of specific conductance, suggesting that a substantial portion of the dissolved components are in an ionized state.

6.5 PHOSPHORUS

The influence of water column stabilities and vertical mixing patterns on phosphorus concentrations are uniquely demonstrated in Baptiste Lake, since the waterbody contains both seasonally and temporarily stratified basins. The seasonal fluctuations in total phosphorus concentration [TP] and orthophosphate concentration [SRP] for the top, middle and bottom sampling depths are illustrated in Figures 25 (south basin) and 26 (north basin). Analytical problems in the provincial goverment laboratory have, however, reduced the usefulness of the TP data between March, 1976 and September, 1976. Furthermore, no field filtrations of [SRP] samples were performed, with the result that phytoplankton and bacteria may have continued to recycle the nutrient until the laboratory analysis was commenced. The [SRP] data should, therefore, be interpreted with caution. Most of the following discussions wil pertain to the data collected between September, 1976 and November, 1977.

During spring turnover of 1977, the surface concentration of [TP] in the south basin was approximately 0.080 mg L^{-1}. This concentration rapidly declined to less than 0.050 mg L^{-1} and remained at such levels throughout the entire growing season. In mid-October, the surface [TP] concentrations increased to the 0.100 mg L^{-1} range and remained at this higher level until freeze-up. An examination of the temperature data (Figure 15) indicates that this large increase in [TP] is coincidental with the development of isothermal conditions in the water column, i.e. fall turnover.

A substantial body of data exists in the literature describing the migration of sediment phosphorus into overlying water during periods of hypolimnetic anoxia (Holdren and Armstrong 1980). In both basins of Baptiste Lake the development of anoxic conditions in the hypolimnia were accompanied by simultaneous increases in hypolimnetic [TP], with maximum concentrations in excess of 1 mg L^{-1} being recorded at the bottom of the south basin.

In the shallow, north basin, a general decline from spring mixing concentrations of 0.060 mg L^{-1} was also observed from May through July. However, substantial increases (~0.100 mg L^{-1}) were observed by August 10, 1977. This basin undergoes a transient period of thermal density stratification (Figure 17), and the rapid increase of surface [TP] appeared to coincide with the incorporation of oxygen-poor water (less than 20% saturated) into the upper water column.

Several other mechanisms may influence [TP] cycling in shallow lakes in summer, including the physical resuspension of sediments during wind mixing, macrophyte senescence (Landers 1982)

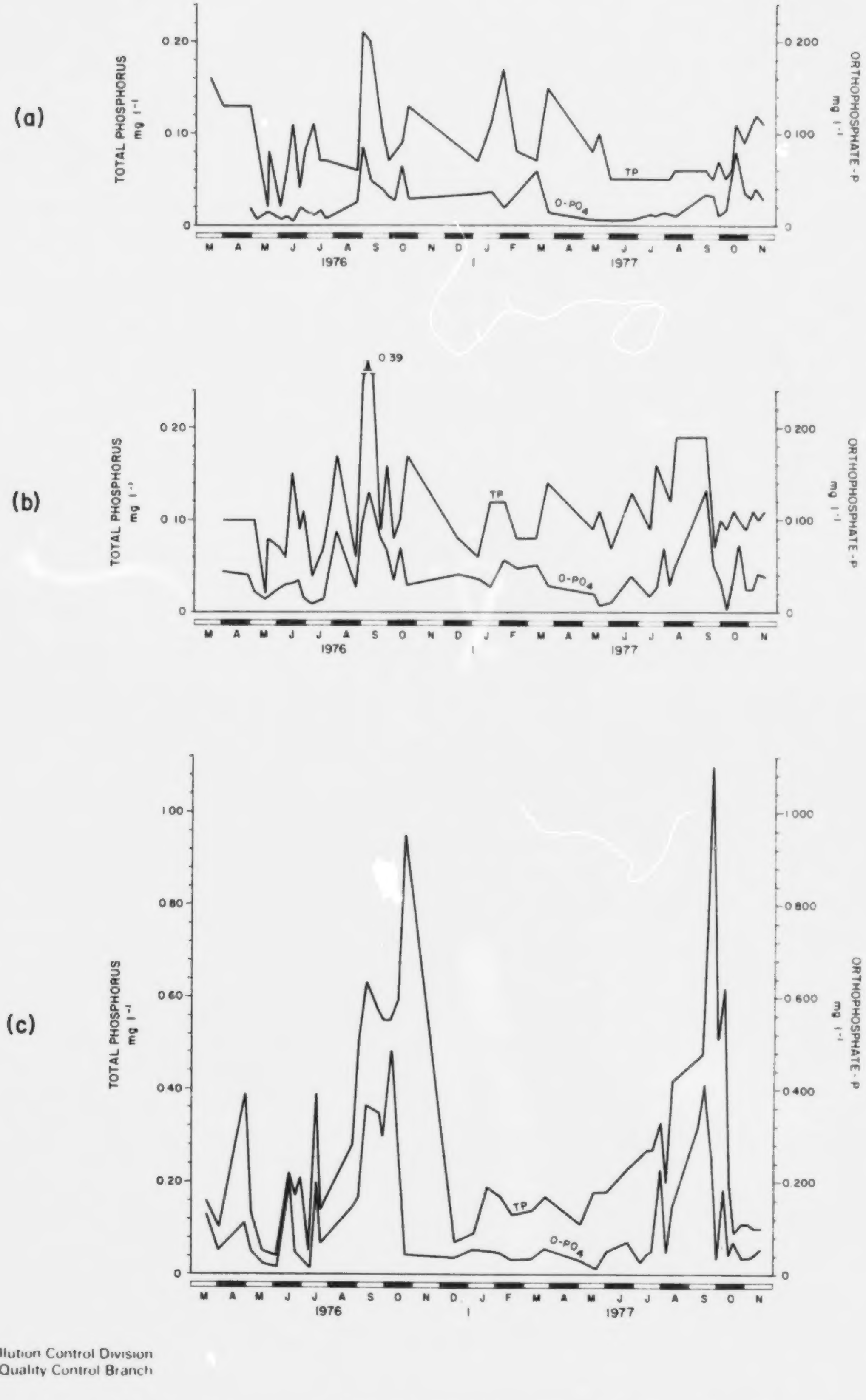

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FIG. 25 SEASONAL FLUCTUATIONS OF TOTAL PHOSPHORUS AND ORTHOPHOSPHATE (SRP) AT (a) 0.5 m, (b) 12.0 m, AND (c) 22.0 m IN BAPTISTE LAKE (SOUTH)

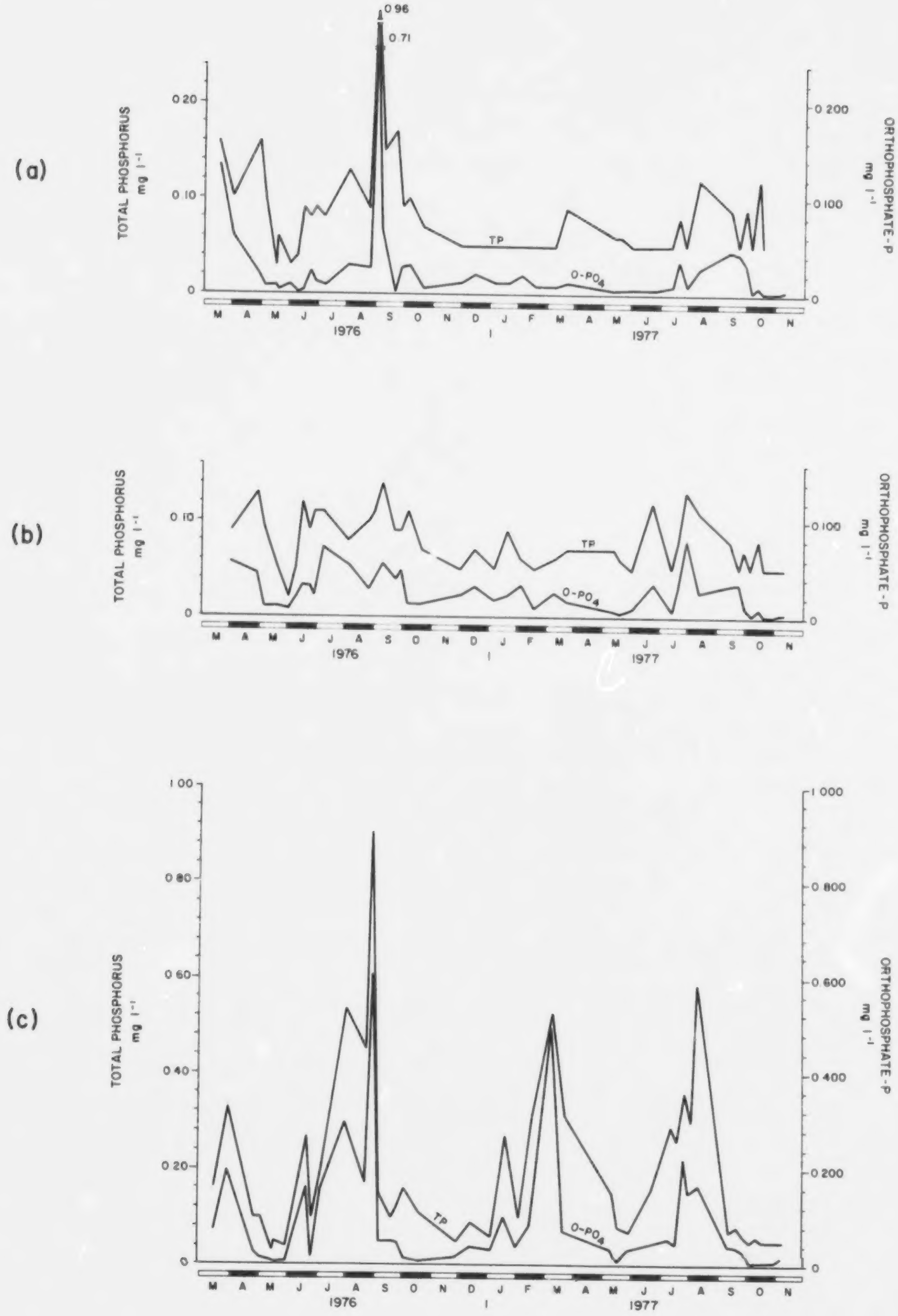

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FIG. 26 SEASONAL FLUCTUATIONS OF TOTAL PHOSPHORUS AND ORTHOPHOSPHATE (SRP) AT (a) 0.5 m, (b) 6.0 m, AND (c) 12.0 m IN BAPTISTE LAKE (NORTH)

and oxic sediment release at higher pH levels brought on by high photosynthetic activity (Anderson 1974(a)).

In order to quantify the significance of these sources a detailed, short-term budget would have to be developed. An important component of this budget would be an estimate of the total mass of phosphorus in the lake. However, the three vertically discrete samples which were taken do not provide enough data for reliable volume-weighting which is essential when a high phosphorus gradient is present.

Seasonal fluctuations in [SRP] at all sites and depths were similar to those of [TP]. Surface concentrations in both basins fell to the 0.005 to 0.010 mg L^{-1} range after ice-out. For the north basin, [SRP] apparently increased to the 0.025 to 0.050 mg L^{-1} range during August and September, in conjunction with the increased [TP], but during the fall declined back to 0.005 mg L^{-1}.

In the south basin, [SRP] remained low throughout the spring and summer (0.015 mg L^{-1}) and only increased after mid-September, following the increases in [TP]. Once again, this probably reflects the influence of hypolimnetic erosion on surface concentrations.

6.6 NITROGEN

The nitrogen cycle in lakes is complex and reflects the interaction of many biological and physico-chemical conditions. The data collected from Baptiste Lake illustrate this complexity by revealing large fluctuations through space and time. Unfortunately, analytical problems in the laboratory (i.e. the ammonia-N analysis was insufficiently sensitive early in the study) and the lack of field filtration to remove plankton from samples complicate the interpretation of the data. The following comments will apply to the period from September, 1976 to November, 1977 (Figures 27 to 30).

Ammonia is produced during the decomposition of proteins or other nitrogenous organic compounds. The oxidation of ammonia to nitrate by nitrifying bacteria is an aerobic process and will continue until oxygen concentrations fall to 0.3 mg L^{-1} (Wetzel 1975). Below this concentration ammonia tends to accumulate or be lost through denitrification.

The ammonia cycle in Baptiste Lake is typical of a stratified eutrophic lake (Wetzel 1975). Prior to destratification in mid-October, 1976, the anaerobic hypolimnion of the south basin served as a large reservoir of ammonia (> 1.50 mg L^{-1}) (Figure 27(c)). As the thermocline became depressed and hypolimnetic erosion commenced, epilimnetic concentrations of ammonia began to rise; this culminated in a massive loading of hypolimnetic ammonia into the water column when overturn was complete. Epilimnetic ammonia concentrations remained high throughout the autumn (Figure 27(a)) but began to decline around the end of December, and fell to below detection (< 0.05 mg L^{-1}) by February. This increase and subsequent decline of ammonia was followed by a progressive increase in nitrate, suggesting that under-ice bacterial nitrification was occurring in the well oxygenated surface waters. Nitrate increased throughout the winter to a maximum concentration of 0.580 mg L^{-1} in early March (Figure 27(a)). This concentration was approximately equal to the sum of nitrite and ammonia concentrations in late

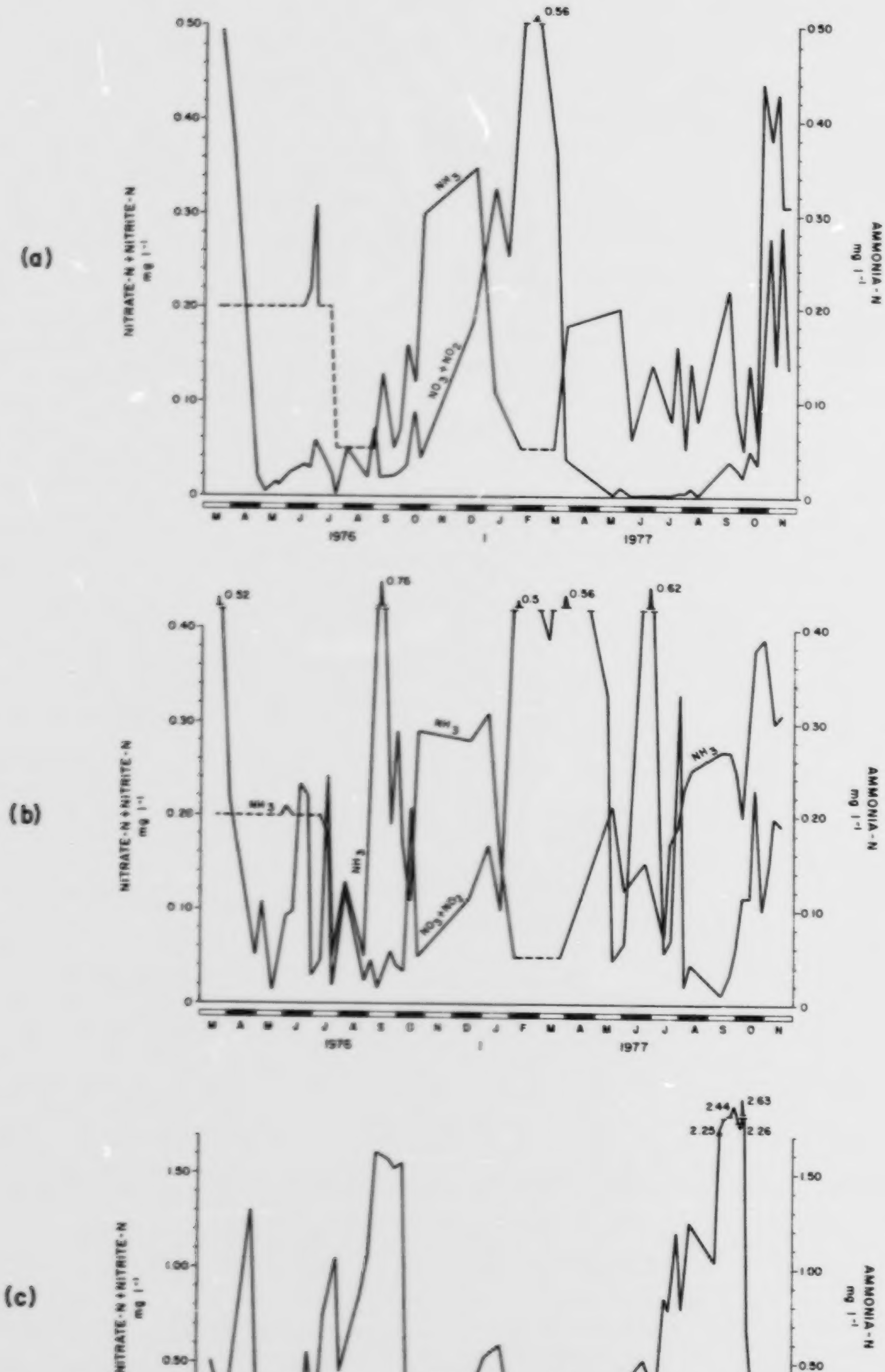

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FIG. 27 SEASONAL FLUCTUATIONS IN AMMONIA AND NITRATE + NITRITE AT (a) 0.5 m, (b) 12.0 m, AND (c) 22.0 m IN BAPTISTE LAKE (SOUTH)

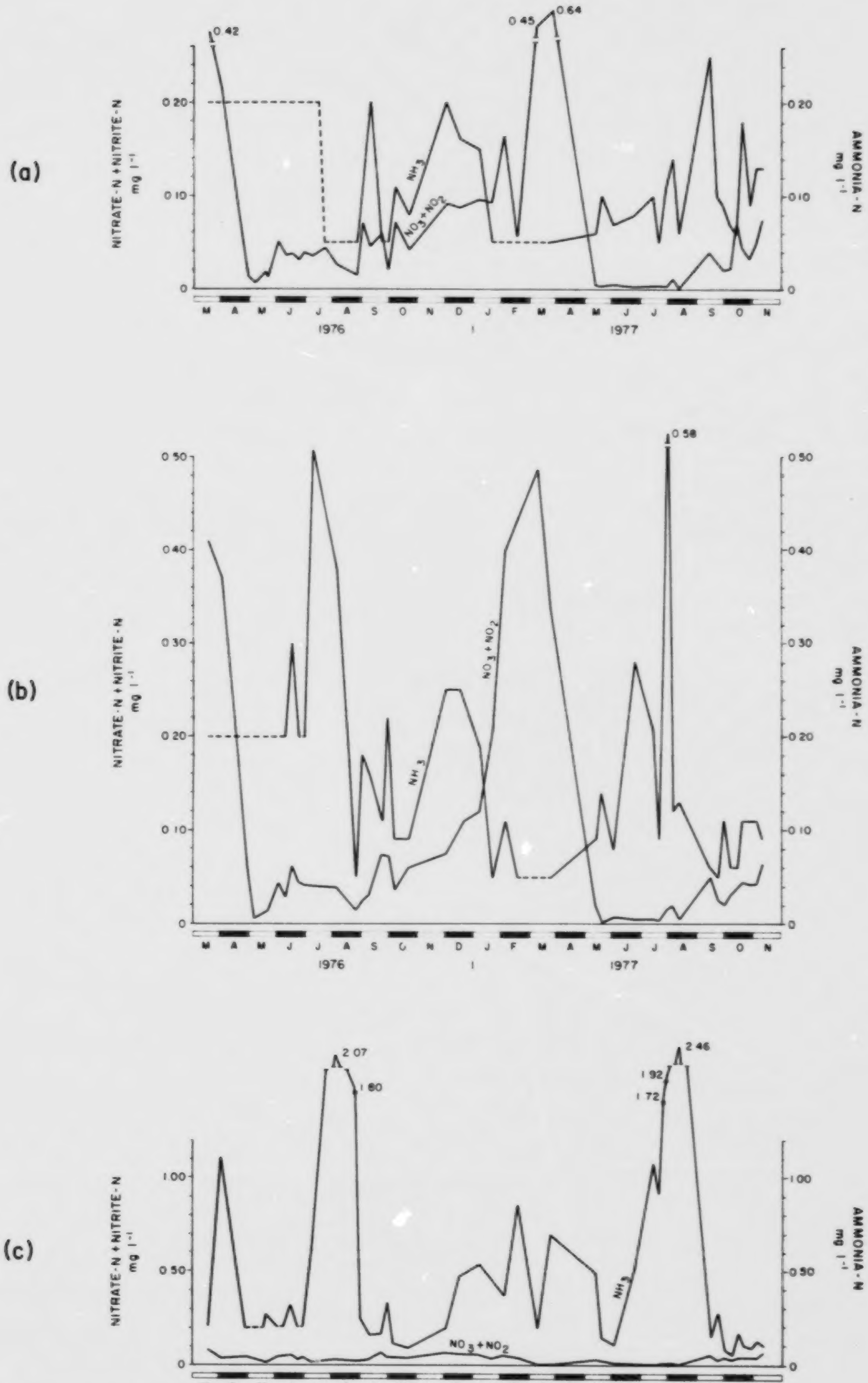

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FIG. 28 SEASONAL FLUCTUATIONS IN AMMONIA AND NITRATE + NITRITE AT (a) 0.5 m, (b) 6.0 m, AND (c) 12.0 m IN BAPTISTE LAKE (NORTH)

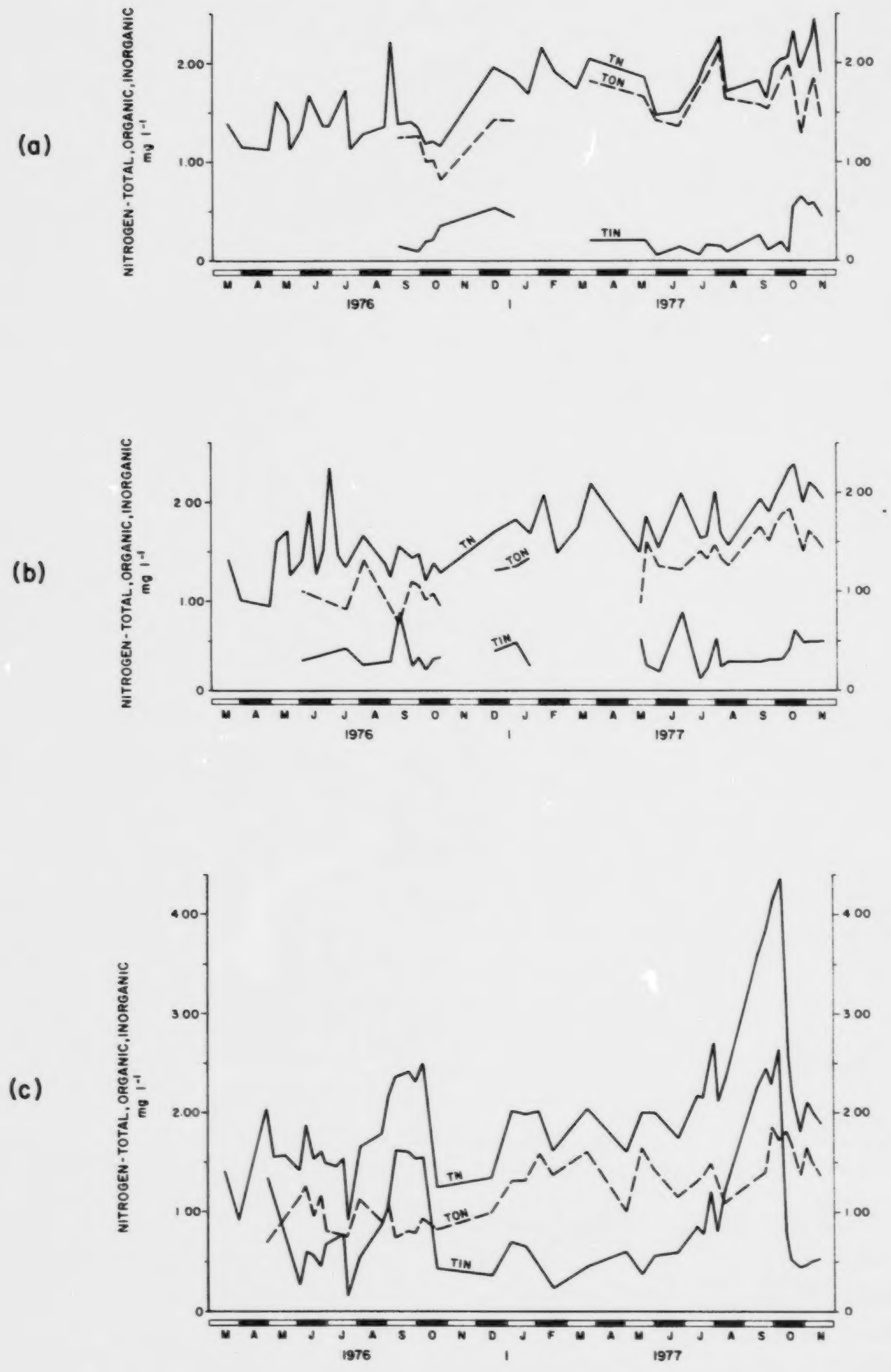

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FIG. 29 SEASONAL FLUCTUATIONS IN TOTAL NITROGEN, TOTAL INORGANIC NITROGEN, AND TOTAL ORGANIC NITROGEN AT (a) 0.5m, (b) 12.0m, AND (c) 22.0m IN BAPTISTE LAKE (SOUTH)

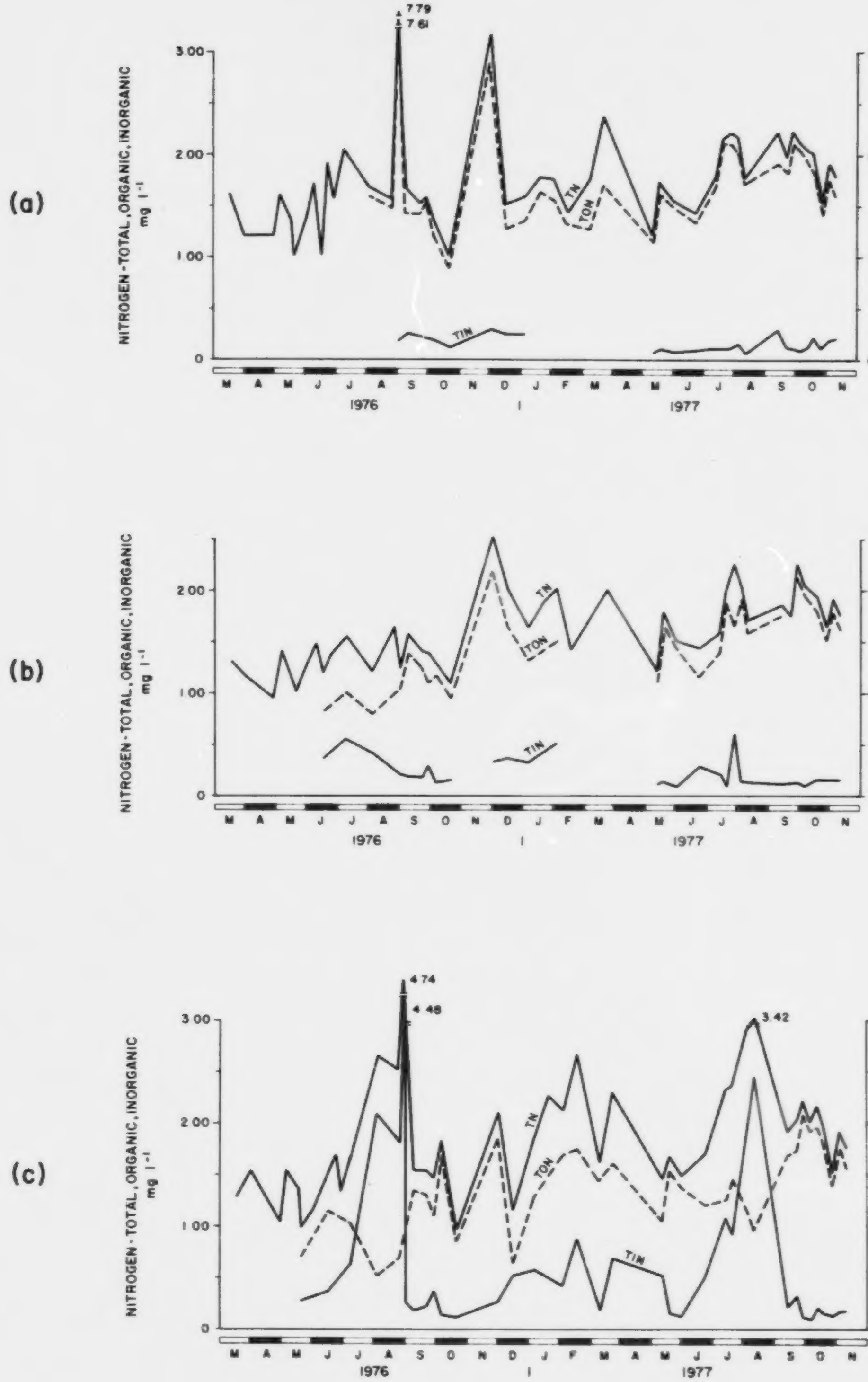

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FIG.30 SEASONAL FLUCTUATIONS IN TOTAL NITROGEN, TOTAL INORGANIC NITROGEN, AND TOTAL ORGANIC NITROGEN AT (a) 0.5 m, (b) 6.0 m, AND (c) 12.0 m IN BAPTISTE LAKE (NORTH)

December, suggesting negligible net losses (ie. denitrification) from the system during winter.

The ammonia cycle during the autumn destratification of 1977 at this site was similar to the previous year. The summer build-up of hypolimnetic ammonia was again high (> 2.0 mg L^{-1}) and evidence of hypolimnetic erosion prior to complete overturn was again observed (Figure 27(b)).

One interesting feature of the hypolimnetic ammonia pool in the south basin of Baptiste Lake is the difference between winter and summer concentrations. There was a much smaller accumulation of ammonia during the winter of 1977 than occurred in either the previous or following summer. Although bottom temperatures were lower in winter (3.5°C) than in summer (6.0°C), the differences are small and might not have altered the rate of release of ammonia from the anaerobic sediments significantly. The absence of autochthonous and allochthonous loadings of organic material in winter and its subsequent decomposition (and utilization of oxygen) seems more likely to explain the large differences between winter (~0.50 mg L^{-1}) and summer (> 1.50 mg L^{-1}) concentrations.

During the spring of 1977, the relative dominance of nitrate over ammonia was quickly reversed in the surface waters. Nitrate was rapidly depleted under the ice, probably the result of the prolific growth of phytoplankton responding to increased photoperiod and light penetration. Further down the water column, this depletion was not evident until later in May (Figure 27(b)). Nitrate concentrations at 0 m were below 0.010 mg L^{-1} from mid-May until mid-August. During this summer growth period, ammonia fluctuated widely in the 0.100 to 0.200 mg L^{-1} range, and may have reflected high volatilization rates as observed on other productive prairie lakes (Murphy and Brownlee 1981).

The patterns of ammonia and nitrate at Baptiste north were generally similar to those of the south basin with the exception that the middle sampling depth at Baptiste north was at 6.0 m and located in well-oxygenated waters throughout the year, whereas the middle depth at Baptiste south (12.0 m) was located in either aerobic and anaerobic waters, depending upon time of year. The net result was that the middle depth at Baptiste north approximated surface conditions, whereas the middle depth at Baptiste south seemed to show characteristics of either epilimnetic and hypolimnetic conditions. For instance, both 1976 and 1977 peaks in ammonia at 12.0 m during August and September predated the surface ammonia accumulation at Baptiste south (Figure 27(b)) suggesting incursions of hypolimnetic water as the thermocline sank. However, during winter this depth was oxygenated and a high degree of nitrification may have occurred, as suggested by the increase in nitrate.

Although we believe that most of the north basin is well mixed, our sampling site was situated at a depth of 14 m and the data suggest that circulation was limited at this site. A shorter, but equally intense summer period of profundal ammonia accumulation was observed during both years at this site. Instability in the water column was evident by early September of both years and so the increase in surface ammonia concentrations occurred much earlier than in the south basin. However, the levels of surface ammonia reached were somewhat lower (~0.20 mg L^{-1}) than in the south basin (0.30 - 0.40 mg L^{-1}) and is probably a reflection of the much smaller mass of hypolimnetic water in which ammonia was accumulating.

The relative proportions of total nitrogen (TN), total organic nitrogen (TON) and total inorganic nitrogen (TIN) are illustrated in Figures 29 and 30. Total nitrogen has been defined as the sum of total Kjeldhal nitrogen and nitrate-nitrite, and total organic nitrogen has been defined as the difference between total kjeldhal nitrogen and ammonia.

The nitrogen pool in Baptiste Lake is dominated by the organic form. In surface waters during the ice-free season, organic nitrogen accounts for over 90% of the total nitrogen (Table 18). The only exception to this pattern occurs in the anaerobic hypolimnon of the south basin (Figure 29(c)) and at the bottom sampling depth of the north basin (Figure 30(c)). In both cases, the accumulation of ammonia is sufficiently large to form the major fraction of the total nitrogen pool. Simultaneous decreases in organic nitrogen at these locations suggest that some degree of mineralization may have occurred.

The surface fluctuations in total nitrogen are erratic (Figures 19(a) and (b)); this may be influenced partly by spatial heterogeneity in phytoplankton (see below) and makes seasonal interpretation difficult. Generally speaking, inorganic nitrogen is highest during fall and winter, and lowest during the growing season. Organic nitrogen, on the other hand, seems to reach its highest levels during the growing season, and this feature probably reflects the peak in annual algal biomass and associated extra-cellular products; the partitioning of these data into dissolved and particulate components would have facilitated more extensive interpretation.

6.7 N:P RATIOS

It has been generally accepted that algae need a relatively fixed atomic ratio of carbon to nitrogen to phosphorus of 106 to 16 to 1 (Vollenwieder 1968; Lee 1973). This corresponds to a nitrogen to phosphorus mass ratio of 7.2 to 1. However, not all fractions of limnetic total nitrogen and total phosphorus pools are bioavailable, and the question of how to calculate N:P ratios remains unresolved (Smith 1979). In view of the predominance of TON in Baptiste Lake, the bioavailability of which is unknown (Murphy and Brownlee 1981), the inorganic component may be more representative. Similarily, the orthophosphate ion is generally regarded as the bioavailable component of the total phosphorus pool (Lean 1973), although it is frequently overestimated by the standard molybdenum blue test and is more correctly described as [SRP] (Rigler 1968).

Thus, mass ratios of inorganic nitrogen to orthophosphate higher than the stoichiometric requirement may be used to indicate that nitrogen is available in excess and that orthophosphate is in a diminished supply relative to the requirement. Conversely, ratios below 7.2 can be used to indicate that nitrogen is in short supply and that orthophosphate is abundant relative to the needs of the algae. Rast and Lee (1978) have used this ratio to distinguish between nitrogen limitation and phosphorus limitation. A similar "ionic" N:P ratio (5:1) was also used by Schindler (1977) to define the limiting nutrient in his whole lake studies in the Canadian Experimental Lakes Area.

Several workers have, however, described the use of total forms of nitrogen and phosphorus to calculate the critical N:P ratio (Sakamoto 1966, 15:1; Smith 1979, 21:1). In order to facilitate comparisons with other lakes; both types of ratios have been calculated for Baptiste Lake during the open water season of 1977.

The inorganic N:P ratios in both basins show a clear decline in the early part of the growing season to minimum values in August and September, suggesting a change from phosphorus to nitrogen-limited growth (Table 19). However, sample handling and analytical problems may have resulted in overestimates of the true orthophosphate, and underestimates in the N:P ratios. The increases in inorganic ratios observed in October were due to the fall turnover which resulted in a re-adjustment of nitrogen and phosphorus concentrations in surface waters. Total N:P ratios remained high (>14.9) throughout the growing season and suggest adequate nitrogen supplies at all times. When compared against empirical phosphorus-chlorophyll a models, the Baptiste chlorophyll a data suggest that phosphorus-proportional growth has occurred (see Section 10.1 below).

6.8 IRON

Total extractable iron was monitored at both stations in Baptiste Lake from October, 1976 to November, 1977 (Figures 31 and 32). According to Wetzel (1975), concentrations of iron in well oxygenated, alkaline surface waters range from 0.050 to 0.200 mg L^{-1}; the waters of Baptiste Lake fall into this category. In

Table 19. N:P ratios in surface waters of Baptiste Lake during 1977.

BN	Inorganic	Total	BS	Inorganic	Total
May 11	16.0	20.1	May 11	103.8	22.2
May 18	20.6	28.7	May 18	34.8	18.8
May 31	15.0	31.1	May 31	12.4	29.4
Jun 22	16.4	28.4	Jun 22	23.8	30.1
Jul 13	11.4	35.9	Jul 13	5.6	36.1
Jul 27	14.1	44.3	Jul 19	16.4	39.7
Aug 03	2.6	43.4	Aug 03	11.5	45.6
Aug 10	2.3	14.9	Aug 10	8.2	28.7
Sep 13	6.3	24.6	Sep 13	7.6	30.6
Sep 21	3.1	39.8	Sep 21	3.6	33.0
Sep 28	3.4	24.9	Oct 05	11.8	40.6
Oct 05	31.0	42.6	Oct 12	1.8	34.4
Oct 12	14.8	17.0	Oct 17	6.8	21.2
Oct 17	13.0	40.5	Oct 26	18.8	21.6
Oct 26	61.5	30.7	Nov 02	19.7	20.5
Nov 02	89.0	38.5	Nov 08	14.6	20.4
Nov 09	41.0	35.9	Nov 16	16.5	17.3

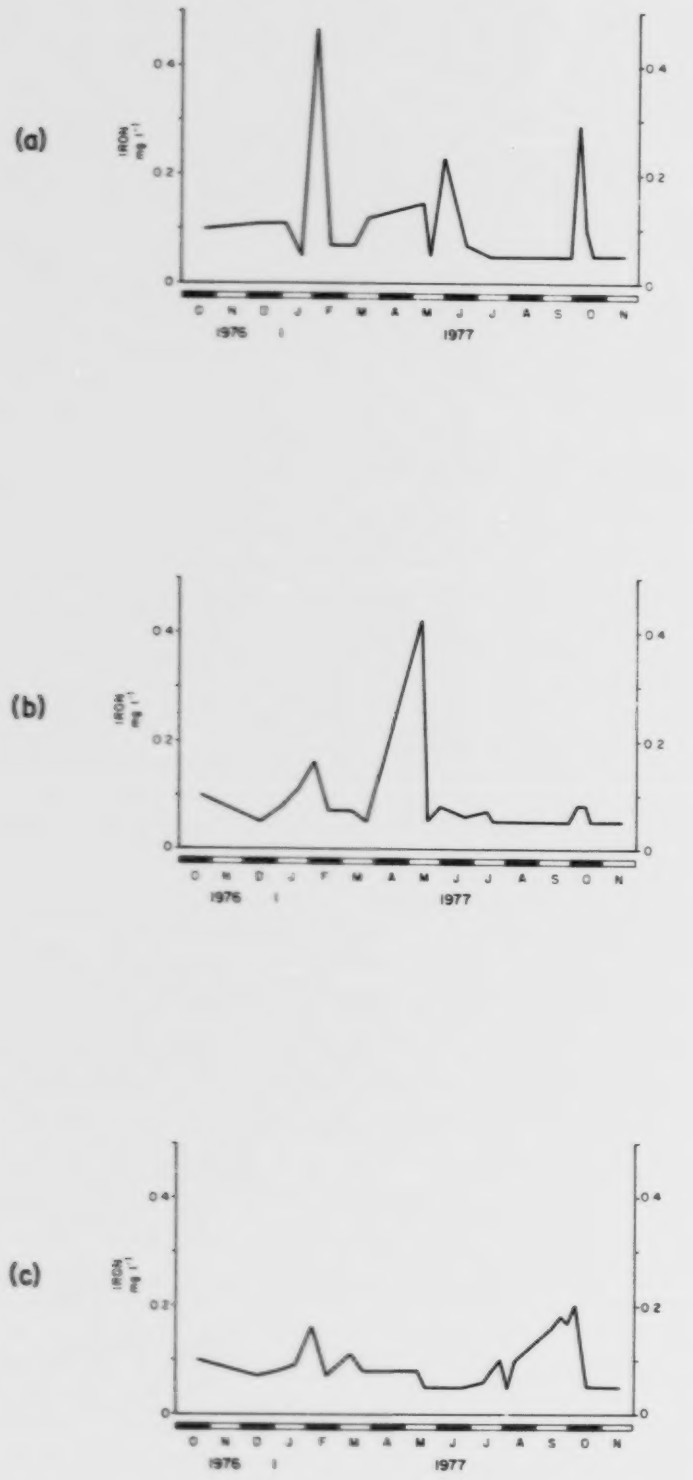

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FIG. 31 SEASONAL FLUCTUATIONS IN IRON AT (a) 0.5 m (b) 12.0 m, AND (c) 22.0 m IN BAPTISTE LAKE (SOUTH)

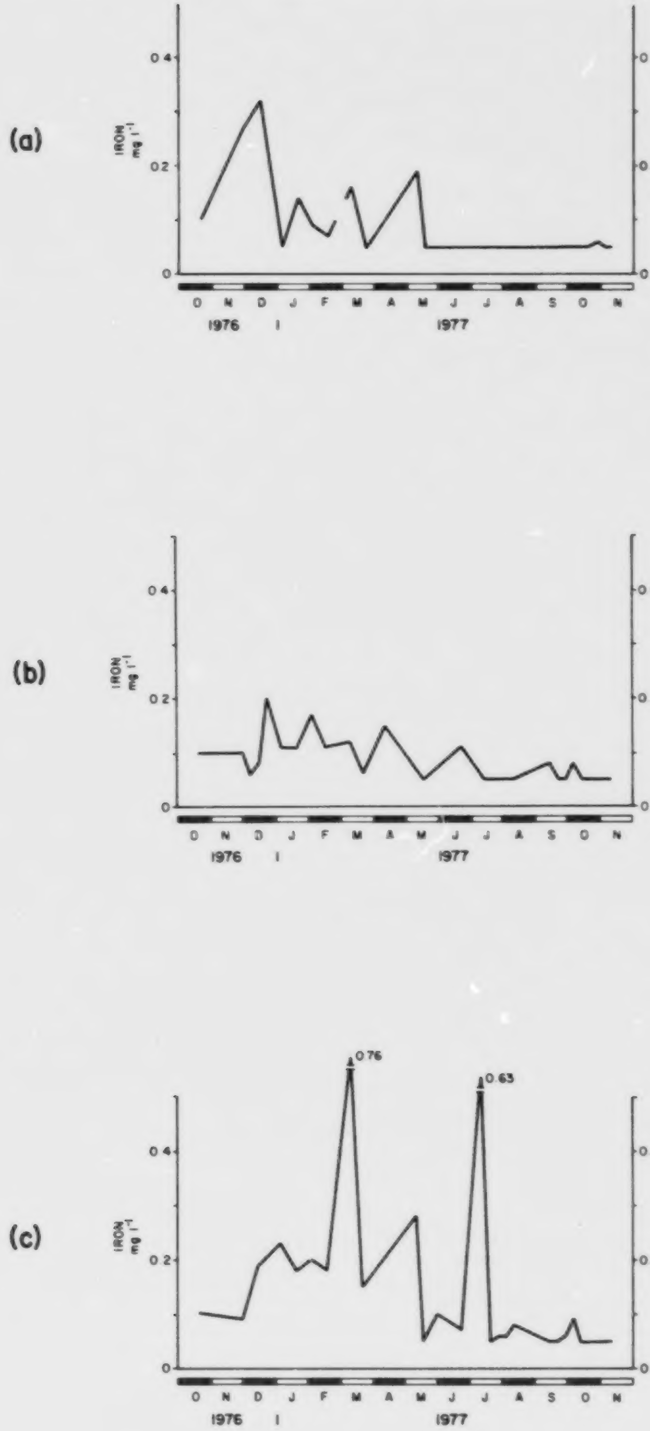

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FIG. 32 SEASONAL FLUCTUATIONS IN IRON AT (a) 0.5 m, (b) 6.0 m, AND (c) 12.0 m IN BAPTISTE LAKE (NORTH)

oxidized conditions, the total iron pool consists primarily of $Fe(OH)_3$, organically complexed iron, and adsorbed iron of seston in particulate form.

Iron concentrations in the surface waters of both basins were generally highest during fall and winter, and were below detection (0.050 mg L^{-1}) for most of the June-September period. The reasons for this cycling are unclear, although phytoplankton requirements during the growing season may directly deplete iron. Golterman (1975) described the role of iron bacteria in mineralizing iron humates, with subsequent sedimentation of $Fe(OH)_3$; this process may be enhanced during the summer due to higher temperatures.

During periods of hypolimnetic anoxia in winter and summer stratification, increased concentrations of iron were observed in both basins. This probably represents the migration of ferrous iron out of the sediments at low redox potentials but in summer may also reflect autochthonous sedimentation of $Fe(OH)_3$, as previously described. As with other constituents, the largest concentration of iron was observed in the north basin during March 1977. Since biotic activities would be minimal at this time, this is further evidence for groundwater infiltration at this site. Winter concentrations in the south basin were much smaller by comparison.

6.9 SILICA

The theoretical role of silica concentrations in controlling the ultimate yield of diatom populations has been extensively documented by Lund et al. (1963). His work in Lake Windermere illustrated the exponential growth of *Asterionella formosa* and subsequent depletion of silica in surface waters, beginning in late winter and culminating in May - June. Diatom populations crashed when silica concentrations fell below 0.5 mg L^{-1}, and sedimentation of cells from the epilimnion increased after the nutrient was depleted. However, Reynolds (1982) proposed that the causative mechanism of such phenomena be attributed to the periodic formulation and collapse of near-surface density gradients (i.e. thermal stratification) and its control on the organisms and their chemical milieu.

Dissolved reactive silica concentrations were monitored at both stations in Baptiste Lake between February and November, 1977 (Figures 33 and 34). In general, Baptiste Lake displays similar fluctuations to Lake Windermere. Diatom growth in the surface water (0 m) of Baptiste Lake commences in March (see below, Figures 39 and 40) and dissolved reactive silica is immediately depleted to concentrations below detection (0.5 mg L^{-1}; Figures 33(a) and 34(a)); the silica concentrations remain at this low level until after the decline of the spring diatom pulse in June. However, a somewhat delayed uptake of silica was observed at the intermediate depths (10.0 m (BN) - 12.0 m (BS)), (Figures 33(b) and 34(c)) and possibly reflects the delayed penetration of favourable light conditions to those depths; these minima were observed in mid-May. A concentration minimum was observed at 20.0 m at Baptiste South in late June. Although light conditions would have been very poor at these depths, the Bacillariophyceae are capable of silica uptake in the dark (Golterman 1975); large numbers of Stephanodiscus astraca cells were observed at 20.0 m in the south basin (not illustrated) prior to the collapse of the epilimnetic populations.

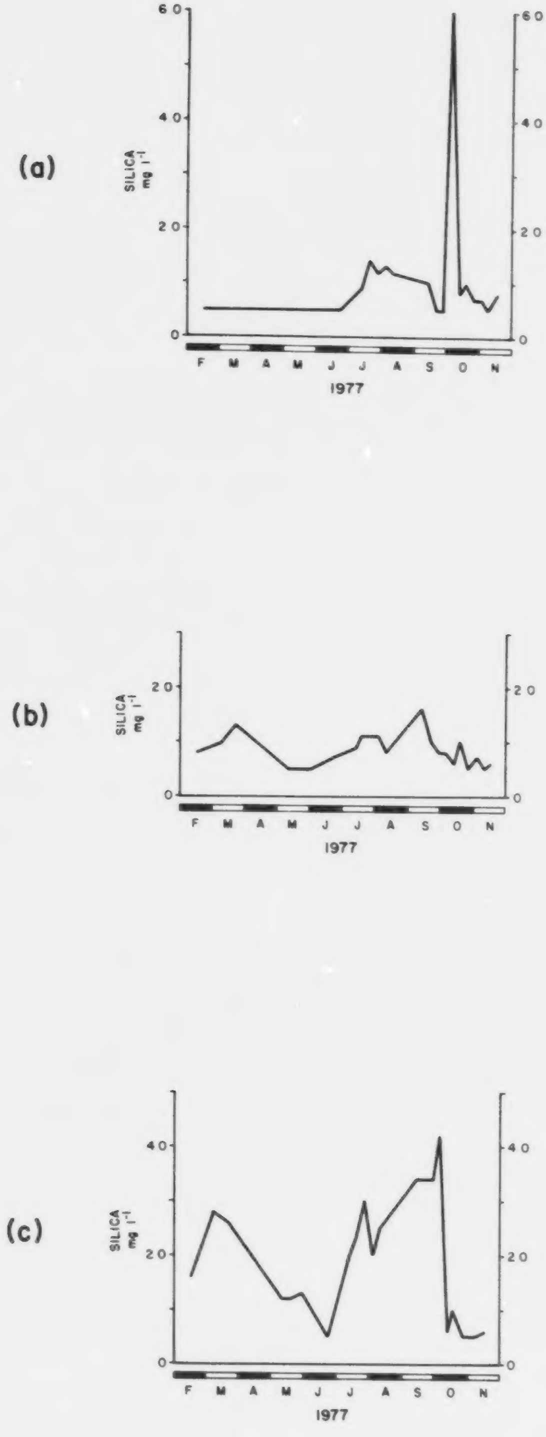

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FIG. 33 SEASONAL FLUCTUATIONS IN SILICA AT (a) 0.5 m, (b) 12.0 m, AND (c) 22.0 m IN BAPTISTE LAKE (SOUTH)

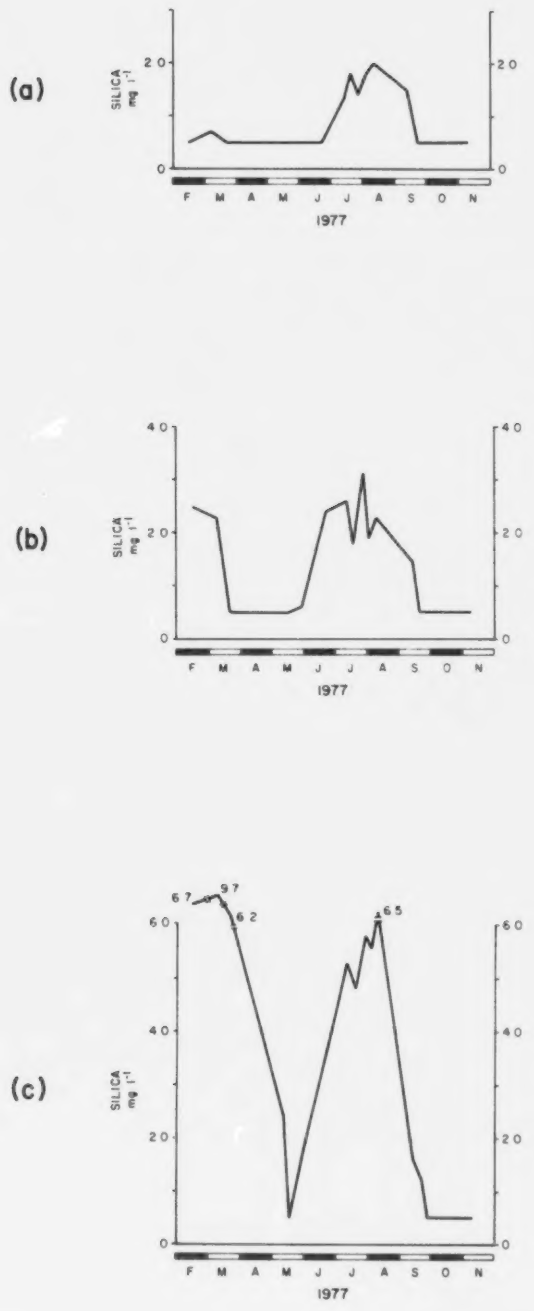

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FIG. 34 SEASONAL FLUCTUATIONS IN SILICA AT (a) 0.5 m, (b) 6.0 m, AND (c) 12.0 m IN BAPTISTE LAKE (NORTH)

No such accumulation of cells was observed at 12.0 meters in the north basin; the silica minimum at this location was, however, correlated in time with the decline of the various nitrogen and phosphorus fractions and may also reflect physical mixing of the water column.

As with many parameters, the highest values of silica occurred during winter at 12.0 m in the north basin (Figure 34(c)), and are probably associated with groundwater; the peak winter concentrations at this location were three times higher than at the bottom of the south basin.

The vertical distribution of silica at BN during the June - August period was correlated with the disappearance of the diatom algae. Silica increased at all depths, and this replenishment may have been derived from several sources: surface runoff, groundwater, decomposition of dead diatom cells, and the diffusion of interstitial silica into overlying waters. Although redox conditions influence the sediment-water exchange of may ionic species, silica diffusion is influenced primarily by temperature (Golterman 1975). The maximum concentrations occurred in mid-August in conjunction with maximum water temperatures. The reduction of silica levels to detection limits (0.5 mg L^{-1}), which occurred by mid-September, was correlated with the onset of the autumn diatom pulse.

A similar replenishment of silica content in the south basin occurred during the June-October period. At the time of overturn in mid-October, the hypolimnetic pool of silica was large enough (Figure 33(c)) to cause a substantial increase in surface concentrations (Figure 33(a)) which had fallen below detection for the latter half of September. However, the autumn diatom pulse was less intense in the south basin than in the north (see Section 7.7), and consequently silica was not totally depleted after mixing.

7.0 TROPHIC STATUS

7.1 PHYTOPLANKTON CHLOROPHYLL a

The annual cycles of chlorophyll a for Baptiste Lake, Island Lake, and Ghost Lake are illustrated in Figure 35; the data are presented in the form of areal water column concentrations (mg m^{-2}), integrated from the surface to 10.0 m at Baptiste North (BN) and 20.0 m at Baptiste South (BS). Since samples from 0 m were not taken during 1976, the data from 0.5 m were extrapolated to the surface to calculate the integrals. The 1977 data, in which 0 m samples were taken, suggests the previous assumption was justified except on rare days of exceptionally heavy blooms (Figures 46 to 49). The discrete chlorophyll a data for all basins are listed in the Appendix in Tables A1 to A4.

The annual cycle of chlorophyll a in the north basin of Baptiste Lake was similar during both years of study (Figure 35(a)). The ice-free seasonal mean concentration was slightly higher during 1976 (314.4 mg m^{-2}, n = 24) than in 1977 (272.6 mg m^{-2}, n = 25). The highest areal concentration observed during the period of study was a bloom which occurred on August 30, 1976 when concentrations reached an estimated peak of 1387.2 mg m^{-2}; however, this integral may be inflated since the 0.5 m data were extrapolated to 0 m. The minimum observed value of 1.9 mg m^{-2} occurred on January 19, 1977.

Winter concentrations throughout the water column were frequently less than 1 mg m^{-3}, reflecting the influence of the poor light conditions which existed under the snow and ice cover.

During late March, chlorophyll a concentrations increased to the 5 mg m^{-3} range immediately below the ice surface due to increased irradiance and the disappearance of the snow cover. Within one or two days of ice-out standing crops had risen to mean concentrations of 28.5 mg m^{-3} (1976) and 15.7 mg m^{-3} (1977) in the 0 - 3.0 m interval (Figures 48 and 49).

The spring increase in 1976 appeared to terminate very near the end of May, whereas in 1977 it was prolonged to mid-June. In addition, the increase in 1976 was more robust having reached chlorophyll a concentrations in the range of 20.0 - 25.0 m^{-3}; by comparison the 1977 event only developed into the 10.0 - 15.0 mg m^{-3} range, possibly the result of cooler water temperatures (Figure 15) and overcast weather which characterized that spring.

The early summer minimum appeared for a two week interval in 1976 with concentrations dropping to the 5.0 mg m^{-3} level in the 0 - 3.0 m interval. The 1977 minimum appeared to be less pronounced (partly masked by a lack of samples in early June) and hovered around the 10 mg m^{-3} level for several weeks. By the end of June in both years the summer growth period was quite evident, with chlorophyll a concentrations having quickly risen to the 40.0 - 80.0 mg m^{-3} range where they remained until the end of September. During the summer growth period (June - August) the bulk of the phytoplankton chlorophyll a was found in the upper 6 meters of the water column. However, during May and September/October when isothermal conditions existed the pigment was evenly distributed throughout the water column.

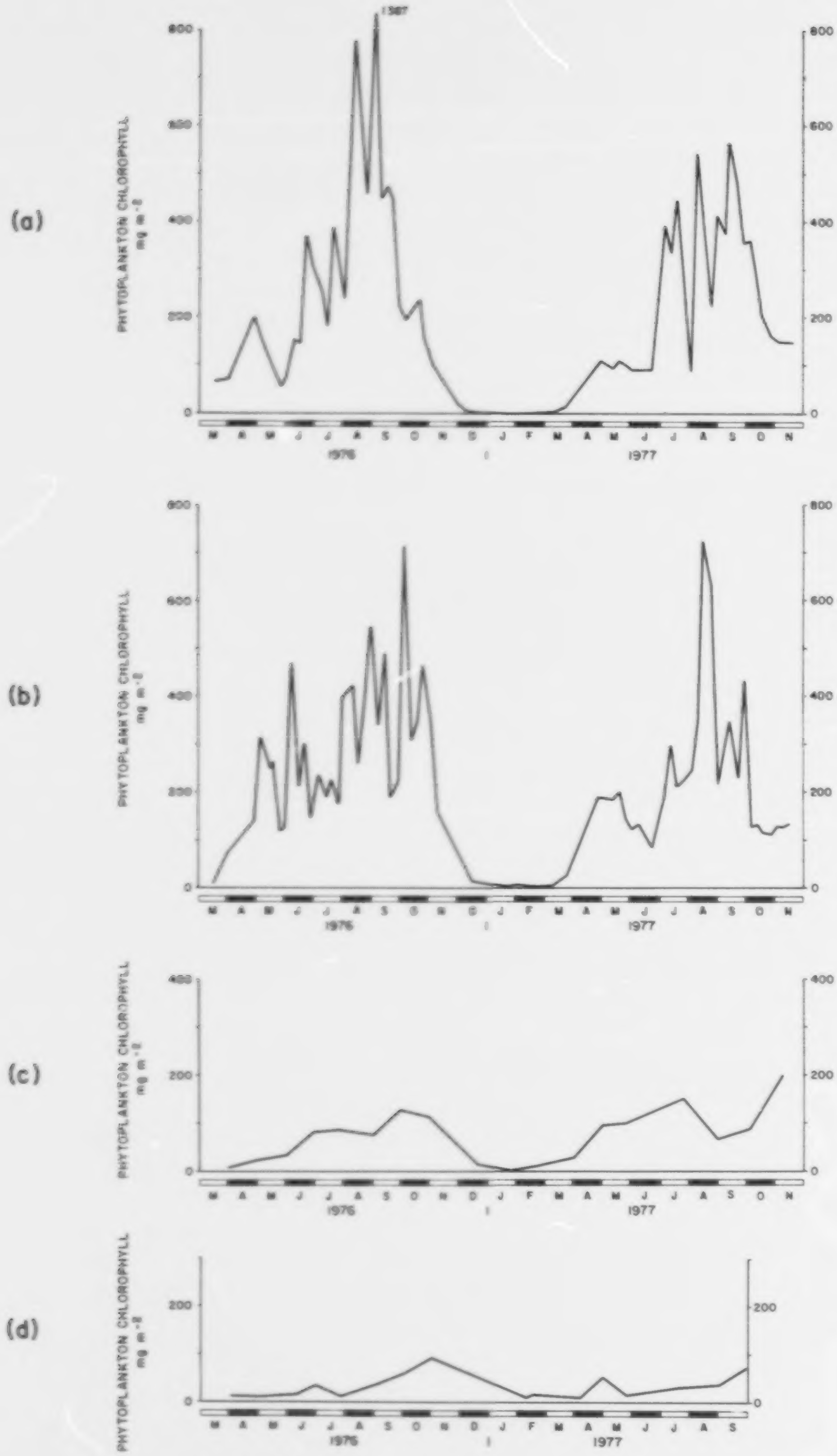

FIG. 35 SEASONAL FLUCTUATIONS IN CHLOROPHYLL CONCENTRATIONS FOR (a) BAPTISTE LAKE (NORTH), (b) BAPTISTE LAKE (SOUTH), (c) GHOST LAKE AND (d) ISLAND LAKE

A number of phenomenal bloom events were encountered during 1976; in all but on one case the north basin displayed a higher standing crop than the south. On August 11, 1976 a concentration of 660.7 mg m^{-3} was observed at the 0.5 m depth. On August 30, 1976 a dense green scum appeared on the surface; samples taken from 0.5 m produced an estimated concentration of 804.6 mg m^{-3}. A smaller bloom occurred on September 21, 1976 with the 0.5 m concentration reaching 140.4 mg m^{-3}. Only one major bloom event was encountered during 1977; this occurred on August 17 and the concentration at 0 m reached 157.0 mg m^{-3}. However, spatial heterogeneity in these surface blooms was significant enough to be visible and so these data points should be interpreted with caution in terms of their representativeness of the whole basin.

Bloom events represent periods of very high growth rates, but also reflect the influence of vertical migration by phytoplankton on the distribution of chlorophyll a. That is, surface concentrations may increase by an order of magnitude during blooms, whereas the total water column content increases by a much smaller degree. The week to week variation in integral concentrations at Baptiste North during June-September, 1976 were much smaller (coefficient of variation = 79%) than the variation in surface concentrations (coefficient of variation = 171%).

The south basin of Baptiste Lake exhibited similar seasonal fluctuations in chlorophyll a (Figure 35(b)), although vertical distributions appear to have been profoundly influenced by thermal stratification. Late winter growth immediately below the ice surface was also evident in this basin during March of both 1976 (approximately 7 mg m^{-3}) and 1977 (approximately 16 mg m^{-3}). Mean concentrations in the 0-3.0 m interval immediately after ice-out reach 20.7 mg m^{-3} in 1976 and 13.1 mg m^{-3} in 1977 (Figures 46 and 47). The incomplete vernal mixing of the basin during 1976 manifested itself with low concentrations at 10 m and 20 m; however, circulation was complete the following spring resulting in a more even distribution of chlorophyll a down the water column.

The spring increases in the south basin were very similar in magnitude and duration to those in the north. In 1976, the early summer minimum appeared at the end of May (mean chlorophyll a at 0-3 m = 7.0 mg m^{-3}) but within four weeks the summer growth period was evident. In 1977, the early summer minimum was somewhat prolonged, as in the north basin, and the summer growth period did not become totally evident until early July. By this time a pronounced vertical stratification in chlorophyll a concentrations was observed, with 10.0 m and 20.0 m samples remaining below 5 mg m^{-3} until the advent of fall mixing.

As noted above, the bloom events were usually of much reduced severity in the south basin. On the dates corresponding to the blooms in the north basin, the peak concentrations observed were 70.7 mg m^{-3} (August 11, 1976), 117.8 mg m^{-3} (August 30, 1976), and 41.3 mg m^{-3} (September 21, 1976). However, on August 17, 1977 the peak concentration of 235.5 mg m^{-3} was higher than that observed in the north. The two basins may therefore bloom out of synchronization, insofar as single sampling sites can reveal.

The 1976 autumn concentrations were sustained at levels greater than 15 mg m^{-3} almost until freeze-up, whereas the 1977 levels were in the 5-10 mg m^{-3} range. The mean areal concentration observed during the 1976 ice-free season was higher (302.7 mg m^{-2}, n = 29) than that observed in 1977 (231.5 mg m^{-2}, n = 27), again following the pattern found in the north basin.

In summary, the north basin produced higher volumetric and areal standing crops during summer than were produced in the south basin, although in spring and fall the situation was reversed (Figure 35). For the open water season as a whole there was apparently no major difference between basins for either 1976 or 1977. The seasonal trends in both basins generally corresponded to the fluctuations in total phosphorus. The chlorophyll a maximum in both basins was always found within the top 3 meters and most frequently near the 0.5 m depth. On days of higher standing crop, the maxima were frequently observed at 0 m.

In a subsequent section dealing with empirical phosphorus-chlorophyll a relationships, various temporal and spatial expressions of whole-lake chlorophyll a concentrations are utilized. These are summarized for 1977 (the year for which most reliable phosphorus data are available) in Table 20.

The seasonal fluctuations in integrated chlorophyll a concentrations for Ghost and Island Lakes are illustrated in Figure 35(c) and (d) respectively. Unlike Baptiste, these waterbodies displayed their highest standing crops during late autumn, and this may reflect the influence of recirculated hypolimnetic nutrients on primary production.

Table 20. Mean chlorophyll *a* concentrations for Baptiste Lake (1977)

Temporal	Spatial	Concentration (mg m^{-3})
Summer	maximum	196.0
Summer (Jun - Aug)	euphotic average	41.2
Annual	epilimnetic average	22.5
Jul - Aug	epilimnetic average	46.8
May - Sep	euphotic average	34.5
Jul - Sep	surface average	55.4
May - Oct	euphotic average	31.7
Spring	maximum	21.7

Table 21. Light extinction as a function of chlorophyll *a*.

Baptiste North	$E = .024\ (B) + 0.784\ (n = 26,\ r^2 = 0.68)$
Baptiste South	$E = .028\ (B) + 0.573\ (n = 35,\ r^2 = 0.57)$
Combined Basins	$E = .027\ (B) + 0.636\ (n = 61,\ r^2 = 0.61)$

Autumn peak concentrations in Ghost Lake were in the range of 20.0 - 25.0 mg m^{-3}, although a strong spring pulse was also detected during the winter of 1977, raising under-ice concentrations to 36.0 mg m^{-3} on March 30. The average open water season concentration for the 0 - 3.0 m interval was 10.0 mg m^{-3} in 1976 and 15.0 mg m^{-3} in 1977 (Figure 50).

The smaller northern basin of Island Lake was the least productive of all basins studied. Autumn concentrations during 1976 exceeded 15.0 mg m^{-3}, but similar data are unavailable for 1977 because of early freeze-up. Average open water concentrations in the 0 - 3.0 m interval for 1976 and 1977 were 4.5 mg m^{-3} and 5.1 mg m^{-3} respectively. A small spring pulse was detected in 1976 but not in 1977. Purple sulphur bacteria were observed on the chlorophyll filters on Sept. 1, 1976 and Sept. 27, 1977. The interferences from bacterial pigments caused negative chlorophyll a values to be produced at depths below 10.0 m.

7.2 PHAEOPHYTIN a

A pronounced seasonality in integral phaeophytin a concentrations was observed in Baptiste Lake during the period of study. The vertical distribution of phaeophytin a is illustrated in Figures 46 to 49; as observed for chlorophyll a, the 1976 concentrations were generally higher than those observed during 1977. The discrete phaeophytin a data for all basins are listed in the Appendix in Tables A5 to A8.

At Baptiste South the highest concentration was observed during the first week of each open water season (1976 = 91.7 mg

m^{-2}; 1977 = 52.1 mg m^{-2}; not illustrated). This may reflect the accumulation of senescent algal material from the late winter pulse which had not yet been grazed because of low zooplankton populations (see below), or which had not been processed by bacteria because of low water temperatures. This interpretation is supported by the fact that high phaeophytin a concentrations observed immediately below the ice during March of both years.

These high spring values declined progressively through May and June to minimum levels in July (1976: less than 20 mg m^{-2}; 1977: less than 10 mg m^{-2}). This decline occurred simultaneously with the rapid increase in chlorophyll and may have reflected a healthy, growing phytoplankton community coupled with higher processing rates of dead material, because maximum epilimnetic temperatures were also observed in July. During late August phaeophytin a began to accumulate in the epiliminon, with a second peak in areal concentration occurring in September (1976: 50 - 60 mg m^{-2}; 1977: 20 - 30 mg m^{-2}). This weakly corresponded with declining water temperatures, but was also associated with a major shift in the composition of the cyanophyte-dominated phytoplankton community from one primarily composed of Anabaena circinalis and an unidentified unicellular form to one dominated by Gomphoshaeria naegelianum, Aphanizomenon flos-aquae, and an unidentified filamentous form (see Section 7.7 and Figure 40). Winter concentrations remained less than 5.0 mg m^{-2} until March.

Phaeophytin a only accumulated in the hypolimnion of the south basin during May and October of each year; summer concentrations at 20.0 m were usually less than 1.0 mg m^{-3}. The accumulation of phaeophytin a at 20.0 m during May of 1976 and 1977 was correlated in time with the wax and wane of two different species at that depth: the unidentified unicellular blue-green and large diatom *Stephanodiscus astraea*. During October of both years the diatom *Melosira granulata* was coincident with phaeophytin fluctuations at these depths, however, *S. astraea* was also present in large numbers in late October, 1976.

Similar seasonal fluctuations were observed at Baptiste North, although there was more scatter in the data. A spring peak was followed by minima in June and July of both years; these minima were followed by rapid increases in August which again corresponded to shifts in cyanophyte community, consisting of declines in *A. circinalis* followed by pulses of *G. naeglianum* and the unidentified filamentous blue-green. There appeared to be no overall difference in the mean open-water euphotic concentrations between basins for 1977 (BS = 20.5 mg m^{-3} (n=27); BN = 20.2 mg m^{-3} (n=24)), although during 1976 the south basin was consistently higher (BS = 45.0 mg m^{-3} (n=29); BN = 34.5 mg m^{-3} (n=23)).

7.3 CHLOROPHYLL a AND LIGHT EXTINCTION

The quality and quantity of light extinction in natural waters is dependent upon four components: (a) pure water, (b) dissolved (and/or colloidal) organic colouring compounds (Gelbstoff), (c) turbidity, due to non-algal particulate matter, and (d) phytoplankton.

Talling (1960) expressed the additive effects of phytoplankton standing crop and all other non-algal sources of light extinction as follows:

$$E_{min} = E_s B + E_q$$

where E_{min} = minimum vertical extinction coefficient (ln units m^{-1})

E_s = incremental increase in extinction per unit chlorophyll a (ln units $m^2\ mg^{-1}$), also known as the self-shading parameter.

B = chlorophyll a concentration of the euphotic zone (mg m^{-3})

E_q = background extinction due to non-algal sources.

In order to define these two components of extinction for Baptiste Lake, regression analyses were performed on extinction coefficients (PAR) and chloropyll a concentrations from corresponding depths in the euphotic zone. The results indicate that significant relationships existed in both basins (Table 21).

The E_s calculated for the combined basin data of Baptiste Lake (.027 ln units $m^2\ mg^{-1}$) is close to that found in other studies: (0.03 - Ministik Lake, Alberta, Hickman 1979(a); 0.03 - Cooking Lake, Alberta, Hickman 1979(b); 0.02 - Lake Windermere, England - Talling 1960; 0.02 - Lake George, Uganda - Ganf 1972).

The background extinction, E_q, is higher in Baptiste North than in Baptiste South, suggesting that the shallow, circulating north basin has a higher content of non-algal particulate material in the water column. The values for E_q (BN = 0.784, BS = 0.573), however, are much lower than those determined by Hickman (1979 a,b) for Cooking Lake (1.53) and Ministik Lake (1.169), both of which have shallow, well-mixed water columns. They are also lower than those detemined in other shallow lakes, such as 2.55 in Lake George (Ganf 1974), 0.85 in Lake Vombsjon (Gelin 1975), and 1.13 in Lough Neagh (Jewson 1977).

The percent contribution of phytoplankton standing crop to light extinction was determined by solving the equation for the combined basin data at low (5 mg m^{-3}) and high (100 mg m^{-3}) chlorophyll a concentrations. The results suggest that at 5 mg m^{-3}, approximately 17% of light extinction is due to phytoplankton, whereas at the higher concentration approximately 75% is due to the phytoplankton. Based upon the mean growing season euphotic concentration of 31.4 mg m^{-3}, 57% of light attenuation is caused by phytoplankton. It is apparent that at low standing crops attentuation is largely due to background factors, whereas at high standing crops it is almost entirely due to algae and therefore $E \rightarrow E_sB$.

7.4 CHLOROPHYLL a CONTENT OF THE EUPHOTIC ZONE

According to Talling (1965, 1973) the euphotic zone (z_{eu}; depth to which 1% of surface penetrating PAR is found) can be approximated by the empirical relationship:

$$z_{eu} = \frac{3.7}{E_{min}}$$

where E_{min} is the extinction of the most penetrating spectral component. As stated previously, no information on light extinction of any band width other than PAR is available on Baptiste Lake, consequently, the euphotic zone depths have been calculated using the relationship:

$$z_{eu} = \frac{4.4}{E_{PAR}}$$

described by Trew et al. (1981) for lakes in the Cold Lake region; this relationship is also similar to that used by Smith (1979) to calculate euphotic zone thickness using PAR.

The range of calculated euphotic zone depths (1.61 - 5.06 m; Tables 22 and 23) is much less than those recently observed in the less productive lakes of the Cold Lake Region (Trew et al. 1981). Furthermore, the mean euphotic zone thickness is smaller in the north basin than in the south, emphasizing both the higher background extinction and higher standing crop at that location.

Talling also proposed that the phytoplankton content of the euphotic zone ΣB_{eu} could be determined as follows:

$$\Sigma B_{eu} = B \times z_{eu}$$

$$\text{or} = B \times \frac{3.7}{E_{min}}$$

assuming that the vertical distribution of phytoplankton is homogenous within the euphotic zone ("B" is the mean chlorophyll _a_ concentration in the euphotic zone). Therefore, the maximum theoretical content of the euphotic zone ΣB_{max} is equal to:

Table 22. Vertical light extinction (E_{PAR}), calculated euphotic zone depths (z_{eu}), average chlorophyll a concentration (B), chlorophyll a content of the euphotic zone (ΣB_{eu}), and total water column chlorophyll a content (ΣB) for Baptiste North (1976).

Date	E_{PAR}	z_{eu}	B	ΣB_{eu}	ΣB
May 19	0.96	4.58	9.5	43.5	85.4
Jun 15	1.48	2.97	21.9	65.0	146.6
Jun 22	2.30	1.91	57.9	110.6	370.9
Jun 28	1.92	1.93	47.7	92.1	311.1
Jul 08	2.28	1.62	49.3	79.9	257.7
Jul 14	1.78	2.47	32.8	81.0	182.4
Jul 20	2.13	1.74	59.9	104.2	389.1
Aug 11	2.08	2.12	62.0	131.4	789.5
Sep 08	1.63	2.27	45.9	104.2	454.6
Sep 15	2.30	1.61	54.9	88.4	475.6
Sep 21	2.30	1.61	61.1	98.4	449.0
Sep 27	1.51	2.45	35.0	85.8	224.0
Oct 05	1.59	2.33	22.2	51.7	195.4

Table 23. Vertical light extinction (E_{PAR}), calculated euphotic zone depths, (z_{eu}), average chlorophyll a concentration (B), chlorophyll a content of the euphotic zone (ΣB_{eu}), and total water column chlorophyll a content (ΣB) for Baptiste South (1976).

Date	E_{PAR}	z_{eu}	B	ΣB_{eu}	ΣB
Apr 28	0.87	5.06	17.7	89.7	141.2
May 06	1.40	3.14	28.2	88.6	316.7
May 19	1.10	4.00	13.5	54.1	263.0
Jun 01	1.06	4.15	11.2	46.3	129.5
Jun 08	1.36	3.23	35.3	114.1	469.7
Jun 22	2.26	1.95	48.6	94.8	303.3
Jun 28	1.23	3.58	20.1	72.1	146.9
Jul 08	1.73	2.54	46.1	117.0	236.0
Jul 20	1.85	2.38	35.0	83.4	224.4
Aug 17	1.81	2.44	37.1	90.4	254.7
Aug 30	1.75	2.51	77.6	195.1	548.8
Sep 08	1.63	2.70	50.1	135.4	339.4
Sep 15	2.13	2.07	32.9	68.2	494.4
Sep 21	1.20	3.67	30.3	111.1	189.8
Sep 27	1.23	3.58	28.7	102.8	223.8
Oct 05	0.92	4.78	37.6	179.9	716.0
Oct 13	0.97	4.56	19.1	86.9	308.9

$$\Sigma B_{max} = B \times \frac{3.7}{E_sB} = \frac{3.7}{E_s}$$

since $E_{min} \rightarrow E_sB$ at high standing crops.

According to these relationships, and using the constant previously developed for PAR (4.4), the maximum theoretical standing crop possible in the euphotic zone of Baptiste North is estimated at 183 mg m^{-2}. The mean maximum for the whole lake would be 170 mg m^{-2}.

These figures are compared with the calculated ΣB_{eu} during the open water season of 1976 for both basins (Tables 22 and 23). The total euphotic zone content has been determined by $\Sigma B_{eu} = B.z_{eu}$ with z_{eu} calculated from 4.4/PAR as described above, and B was taken as the average of chlorophyll a concentrations at 1.0, 2.0, and 3.0 m.

The data indicate that the average euphotic zone content of chlorophyll a at Baptiste North was 87.4 mg m^{-2}, or approximately 48% of the theoretical upper limit. One should note, however, that the absence of E_{PAR} data at BN during late July and August has undoubtedly biased ΣB_{eu} to the lower spring/autumn values. At Baptiste South, the average content was 101.8 mg m^{-2} or 65% of the theoretical upper limit at that site. Based on these data, the average euphotic zone content for the whole lake is 94.6 mg m^{-2} or 56% of the maximum possible. Finally, it is worth noting that the calculated euphotic zone contents of both basins were much less (approximately 30%) than the total water column content, i.e. 70% of the calculated chlorophyll a content was below the euphotic zone. Again one should note the inherent problems associated with the integration of discrete data points through large intervals of the water column, i.e., from 6.0 to 10.0 m or 10.0 m to 20.0 m. The presence of a single large concentration of chlorophyll a at 10.0 m, such as one might encounter in an algal plate, inflates the total water column integral significantly.

According to Talling et al. (1973), the maximum integral chlorophyll a concentrations thought to occur in nature are in the range of 180-300 mg m^{-2} with self-shading indicated as the major limiting factor. Megard et al. (1979) suggest that the variability in the partioning of light extinction between algal and non-algal components may have a major influence on whether or not these maxima are attained for a given nutritional status.

7.5 SPATIAL CHLOROPHYLL a DISTRITBUTION

In addition to the routine chlorophyll a observations at the two main sampling sites, a number of regular, lake-wide surface chlorophyll a distribution surveys were undertaken. The purpose of these surveys was to provide a frame of reference for the sampling being conducted at the principal stations by investigating spatial heterogeneity. The lake was divided into a grid consisting of 53 equal sized quadrants with one sampling site located in each quadrant (Figure 36).

Fifty-three samples were taken at the 0 m depth on five dates during 1976: May 31, June 16, July 5, August 15 and September 7. The data from these surveys have been superimposed on outline maps of the lake, and areas of similar concentration have been

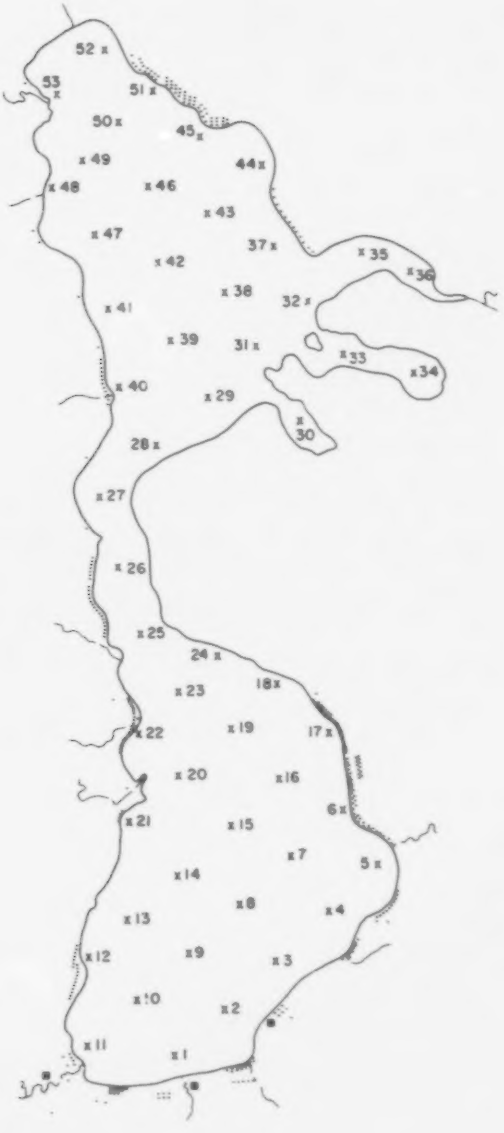
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FIG. 36 SAMPLING GRID FOR SURFACE CHLOROPHYLL AND BACTERIAL SURVEYS IN BAPTISTE LAKE

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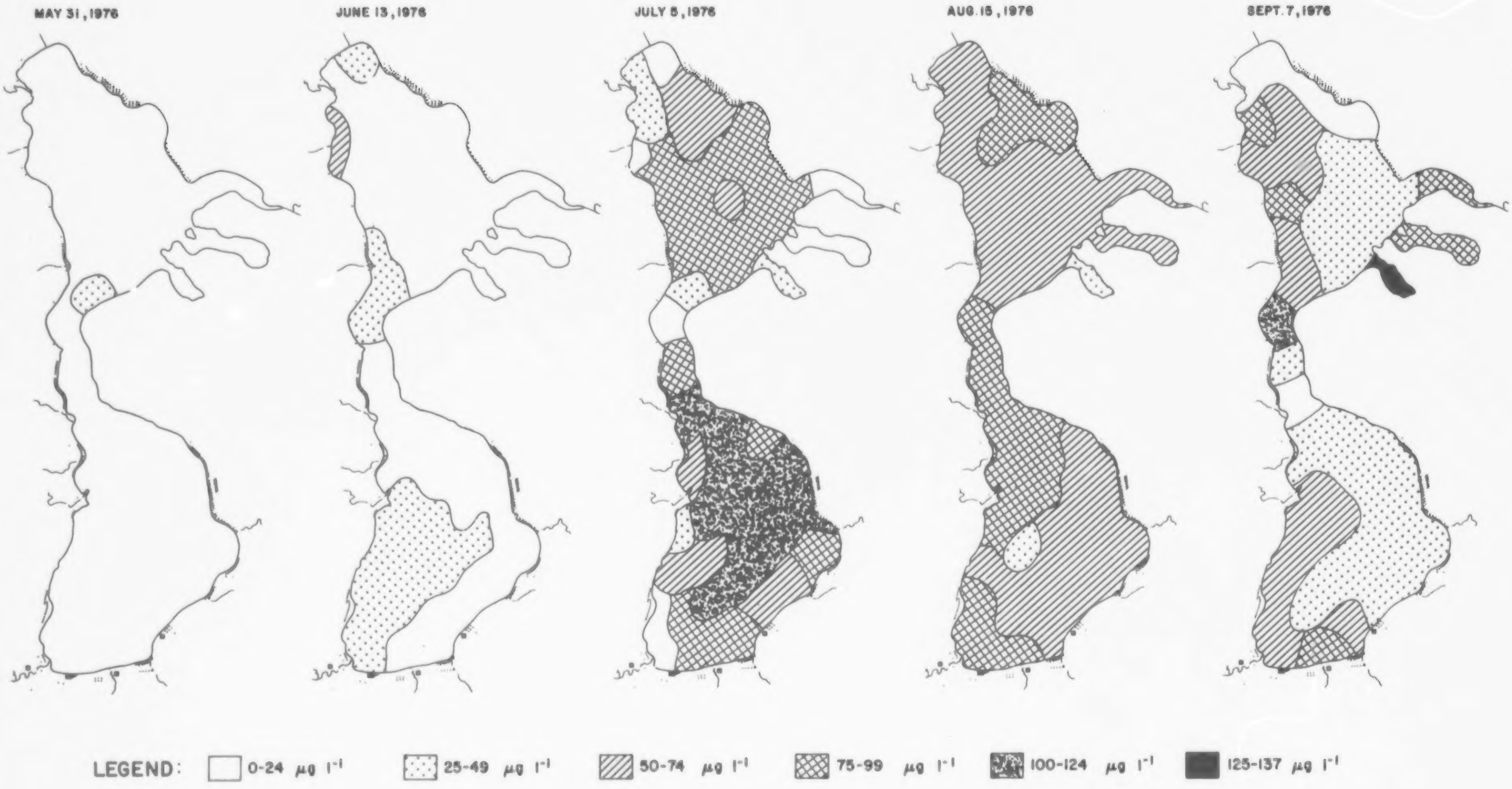
MAY 31, 1976
JUNE 13, 1976
JULY 5, 1976
AUG. 15, 1976
SEPT. 7, 1976
LEGEND:
0-24 μg l⁻¹
25-49 μg l⁻¹
50-74 μg l⁻¹
75-99 μg l⁻¹
100-124 μg l⁻¹
125-137 μg l⁻¹

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FIG. 37 SURFACE DISTRIBUTION (0 m) OF CHLOROPHYLL IN BAPTISTE LAKE

shaded accordingly (Figure 37). Patch development became most intense in summer as the standing crop peaked. The shallow, north basin displayed a higher coefficient of variation than the deeper, south basin on all sampling dates except August 15, 1976 (Table 24). The highest variability in the surface distribution of chlorophyll a ranging from 16.2 mg m^{-3} to 136.5 mg m^{-3} was observed in the north basin on September 7, 1976 (coefficient of variation (C.V.) = 56.4%); the largest proportional range (2.9 mg m^{-3} to 90.3 mg m^{-3}) was measured in the same basin on July 6, 1976. Clearly, the selection of a single sampling site on either sampling date would have provided grossly inadequate data with respect to the whole basin.

A general correlation between increasing variance and increasing standing crop was noted which is a common characteristic of biological communities. An analysis was conducted to determine the number of samples required to give a standard error of ± 10% of the mean. According to Elliot (1977), the number of samples required can be calculated by the following formula when the distribution is unknown:

$$n = \frac{s^2}{D^2 \cdot \bar{x}^2}$$

where n = required sample size

s^2 = sample variance

D = ratio of standard error to the arithmetic mean

$\bar{x}$ = sample mean

Table 24. Chlorophyll a survey statistics

Basin	Date	n	s^2	x	C.V.
Baptiste North	31-5-76	28	17.7	13.1	32.1
Baptiste South	31-5-76	25	5.0	12.3	17.9
Baptiste North	13-6-76	28	132.3	21.7	53.0
Baptiste South	13-6-76	25	41.1	22.7	28.2
Baptiste North	5-7-76	28	840.3	57.7	50.3
Baptiste South	5-7-76	25	972.5	86.2	36.2
Baptiste North	15-8-76	27	102.4	66.3	15.3
Baptiste South	15-8-76	25	190.4	70.2	19.7
Baptiste North	7-9-76	28	967.8	55.1	56.4
Baptiste South	7-9-76	25	260.8	44.5	36.4

Table 25. Required number of samples for ± 10% standard error

Date	BN	BS
May 31, 1976	10	3
Jun 13, 1976	28	8
Jul 05, 1976	25	13
Aug 15, 1976	2	4
Sep 7, 1976	32	13

The estimated sampling requirements based on the five surveys are presented in Table 25.

These and the vertical distribution data suggest that an arbitrary top, middle, bottom type of sampling approach at a few locations in a lake may be inadequate for the quantitative assessment of the whole-lake chlorophyll _a_ concentration. By implication the same concern applies to the whole-lake determination of phosphorus content, since a certain portion of the total phosphorus pool is tied up in phytoplankton cells.

7.6 PHYTOPLANKTON CELL COUNTS

The annual fluctuations in total phytoplankton cell counts for the four study basins are illustrated in Figure 38; the data are presented in the form of water column integrals (cell m^{-2}) as described above for chlorophyll a. The daily profiles of phytoplankton cell counts are illustrated in Figures 46 to 51. The discrete cell count data for all basins are listed in the Appendix in Tables A9 to A12.

The interpretation of cell count data in terms of phytoplankton biomass is complicated by the variability in cell sizes within and between species. Cell volumes have consequently been estimated for all species in order that comparisons with cell counts and chlorophyll a could be made. These estimates are listed by basin and species in the Appendix in Table A13.

In general, the integral cell counts for Baptiste Lake followed the annual pattern observed for integral chlorophyll a, although there were marked differences in vertical distribution. In the south basin the peak concentration during the spring pulse of 1976 (353×10^9 cells m^{-2}) was observed on April 28, 1976. This was followed by a decline to a minimum concentration of 166×10^9 cells m^{-2} on May 17, 1976, a value which was not reached again until November of that year. By mid-June the summer growth phase was evident with standing crops reaching 1490×10^9 cells m^{-2} on June 8; concentrations remained in the $500 - 1000 \times 10^9$ cells m^{-2} range until early October when the most rapid period of decline was observed. Concentrations in November were less than 100×10^9 cells m^{-2}. The minimum value for the

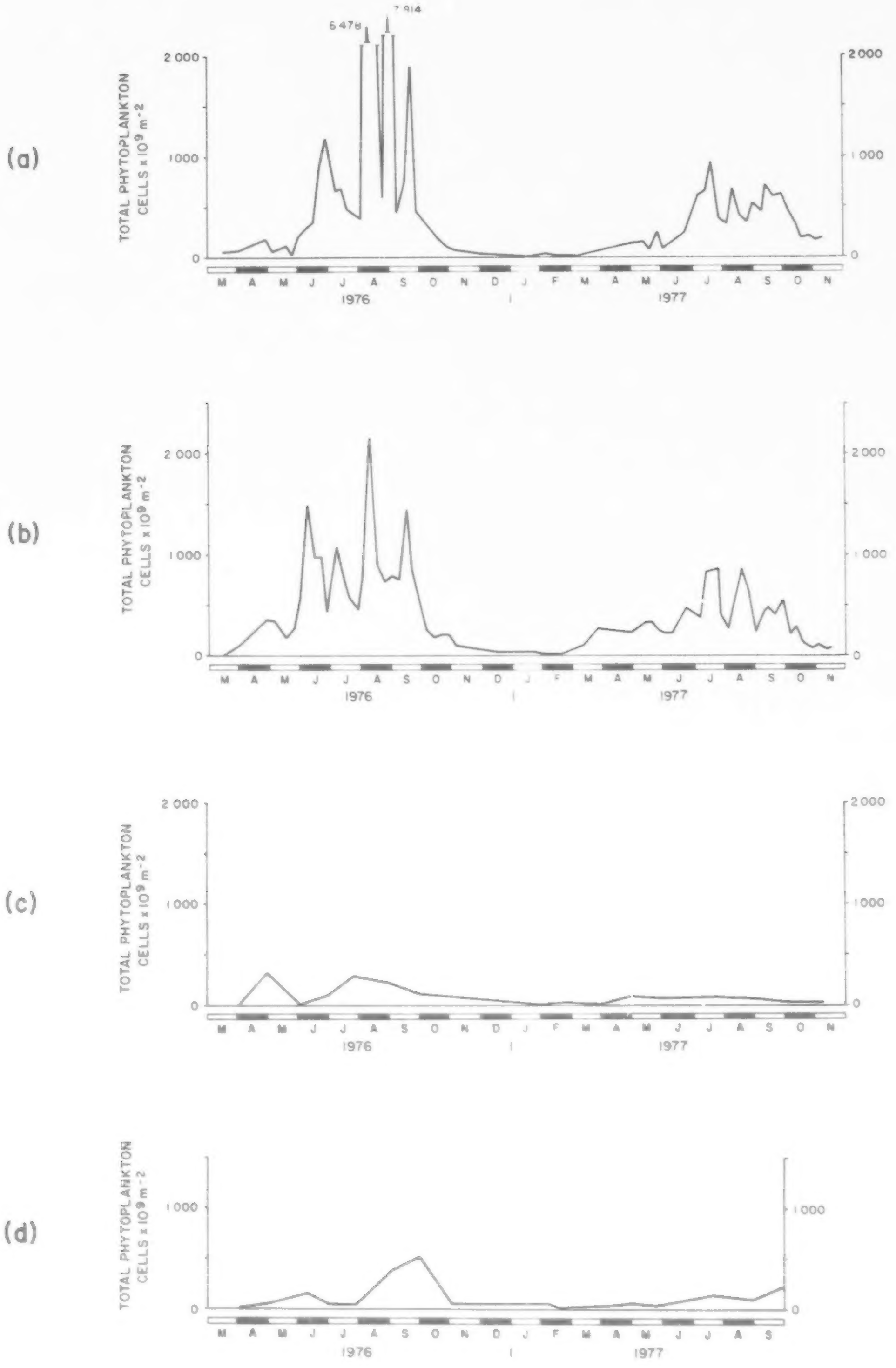

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FIG. 38 SEASONAL FLUCTUATIONS IN PHYTOPLANKTON CELL NUMBERS FOR (a) BAPTISTE LAKE (NORTH), (b) BAPTISTE LAKE (SOUTH), (c) GHOST LAKE AND (d) ISLAND LAKE

entire period of study was 5.1 x 10^9 cells m^{-2} recorded on Feb. 16, 1977. During the remainder of 1977 the seasonal pattern was similar to that of 1976 although the summer growth period was not clearly evident until early July.

The seasonal fluctuations in the north basin were similar to those observed in the south basin. As described previously, this basin displayed several intense blooms during the summer of 1976, and on both August 11/76 and August 30/76 the concentration of algal cells exceeded 10^7 cells per mL at 0 m.

The highest concentration of cells in Island Lake was observed in autumn, tracking the seasonal pattern described for chlorophyll a. However, in Ghost Lake there was a departure from the observed chlorophyll a pattern, with low numbers of cells observed during autumn.

There was a discrepancy between the vertical distribution of chlorophyll a and cell counts in the hypolimnion of the south basin. The profiles illustrated in Figures 46 and 47 clearly demonstrate that large number of phytoplankton cells were observed at 20.0 m during the May - July period of both years, although chlorophyll a was negligible at those depths. These observations suggest that either the phytoplankton were not living autotrophically at those depths and hence did not produce extensive photosynthetic pigments, or that we were recording senescent, sedimenting material from the epilimnion. In any case, the use of chlorophyll a as an index of phytoplankton biomass at such depths may be misleading. We therefore tested for the relationship between these two indices by regressing chlorophyll a (mg m^{-3}) against

Table 26. Relationships between chlorphyll a (mg m^{-3}) and cell counts (x 10^9 cells m^{-3}).

Basin	Depth	Equation	r^2	n
BS	0 m	$\log_{10}$ [chl] = 0.69 $\log_{10}$ [cells] + 0.34	0.76	27
BS	0.5 m	$\log_{10}$ [chl] = 0.53 $\log_{10}$ [cells] + 0.60	0.68	55
BS	1.0 m	$\log_{10}$ [chl] = 0.59 $\log_{10}$ [cells] + 0.47	0.72	56
BS	2.0 m	$\log_{10}$ [chl] = 0.55 $\log_{10}$ [cells] + 0.57	0.69	53
BS	3.0 m	$\log_{10}$ [chl] = 0.46 $\log_{10}$ [cells] + 0.46	0.55	53
BS	10.0 m	$\log_{10}$ [chl] = 0.65 $\log_{10}$ [cells] + 0.16	0.41	55
BS	20.0 m	$\log_{10}$ [chl] = 0.11 $\log_{10}$ [cells] + 0.32	0.01	42
BN	0 m	$\log_{10}$ [chl] = 0.75 $\log_{10}$ [cells] + 0.23	0.70	25
BN	0.5 m	$\log_{10}$ [chl] = 0.54 $\log_{10}$ [cells] + 0.60	0.62	47
BN	1.0 m	$\log_{10}$ [chl] = 0.57 $\log_{10}$ [cells] + 0.55	0.68	49
BN	2.0 m	$\log_{10}$ [chl] = 0.57 $\log_{10}$ [cells] + 0.57	0.71	50
BN	3.0 m	$\log_{10}$ [chl] = 0.52 $\log_{10}$ [cells] + 0.66	0.66	50
BN	10.0 m	$\log_{10}$ [chl] = 0.72 $\log_{10}$ [cells] + 0.18	0.58	45

cell counts (x10^9 cells m^{-3}) for individual depths in both basins; the results are summarized in Table 26.

In surface waters there was generally a good agreement between these two indices, but there was no significant relation between the two at 20.0 m.

7.7 PHYTOPLANKTON TAXONOMY

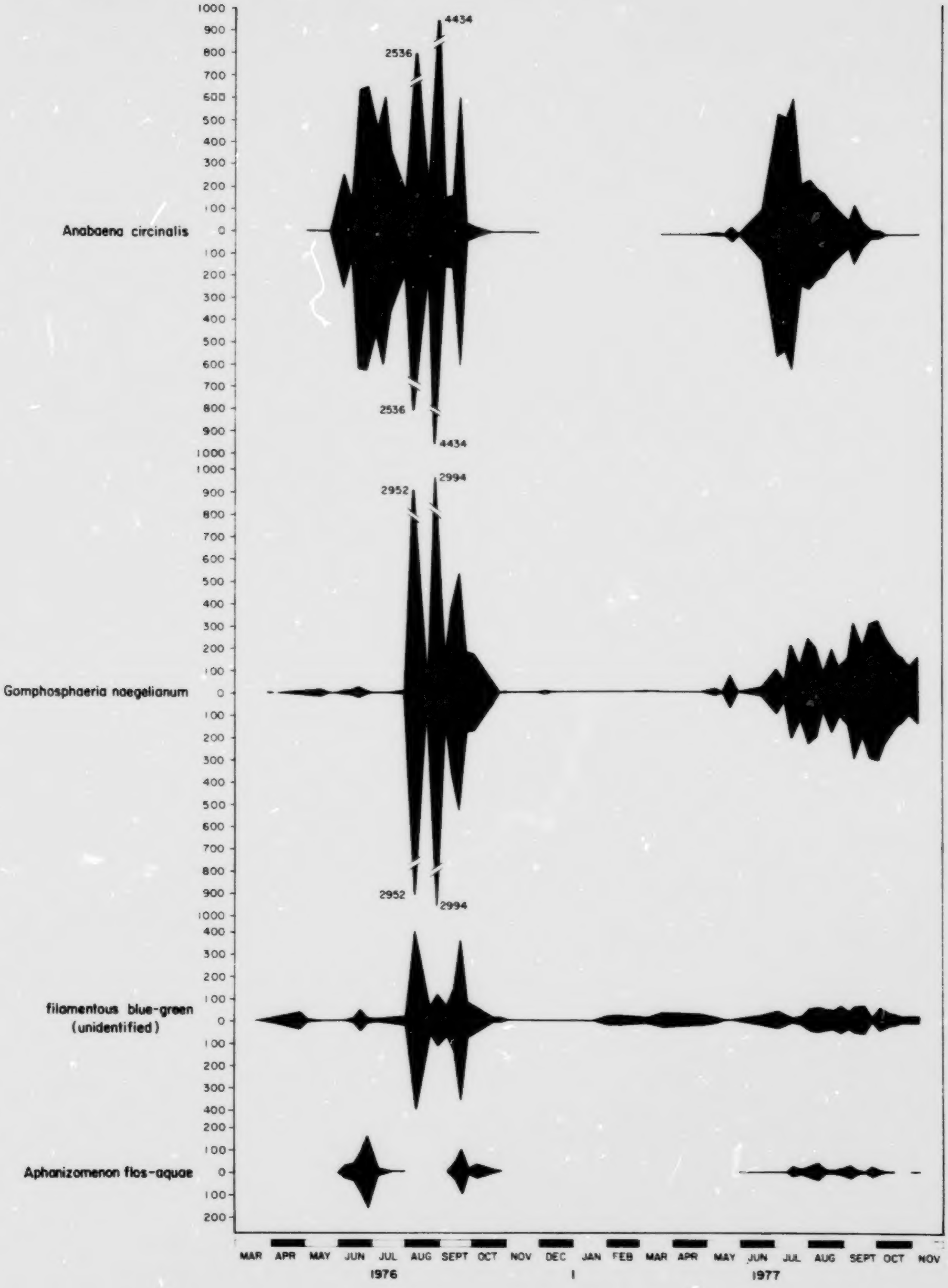
Because of the highly variable relationships between cell counts and cell volumes (within and between species), both parameters are included in the following discussion.

The phytoplankton flora of the study basins have been identified and classified into six divisions: Cyanophyta, Bacillariophyta, Chlorophyta, Pyrrhophyta, Crytophyta, and Chrysophyta; at least 30 species were identified in each basin (Table A13, showing cell volumes). The annual fluctuations in cell counts of the dominant species within these divisions are illustrated in Figures 39 - 42.

The average cell counts and cell volumes for the open-water season and percent compositions are presented by division in Tables 27 and 28 respectively. The interpretive problems that arise with cell count/volume data are illustrated in this table. The Cyanophyta were numerically the most dominant group in all four basins, representing over 90% of the mean integral cell counts in both years with the exception of Ghost Lake in 1977. However, in terms of calculated cell volumes the picture becomes quite different. For instance, at Baptiste South in 1977 the blue-greens represented 90.9% of integral cell counts but only 16.5%

Table 27. Phytoplankton cell count (x10^9 m^{-2}) and cell volume (cm^3 m^{-2}) open-water means

	Baptiste (North)		Baptiste (South)		Ghost		Island	
	1976	1977	1976	1977	1976	1977	1976	1977
Cyanophyta								
Cell Number	1108.00	368.00	625.00	325.00	132.00	32.80	161.00	103.00
Cell Volume	51.00	17.70	26.10	12.40	7.18	3.79	4.19	1.71
Chlorophyta								
Cell Number	2.01	6.61	1.55	4.82	0.90	1.88	1.23	4.49
Cell Volume	0.29	2.17	0.36	1.75	0.19	0.65	0.23	2.46
Bacillariophyta								
Cell Number	9.95	25.00	7.64	20.50	11.20	8.80	4.28	1.12
Cell Volume	7.18	41.10	10.50	36.40	10.40	6.64	9.42	.59
Chrysophyta								
Cell Number	5.34	0.25	1.01	0.14	1.24	10.20	0.57	1.33
Cell Volume	0.97	0.12	0.66	0.18	0.78	6.34	0.36	0.57
Pyrrhophyta								
Cell Number	0.12	0.13	0.15	0.92	0.02			0.10
Cell Volume	5.55	10.30	8.33	21.40	0.84			0.57
Cryptophyta								
Cell Number	5.72	3.15	3.91	6.25	1.24	4.93	2.30	4.33
Cell Volume	1.82	2.15	1.82	2.91	0.75	2.51	0.72	2.18


Anabaena circinalis
Gomphosphaeria naegelianum
filamentous blue-green (unidentified)
Aphanizomenon flos-aquae
1000
900
800
700
600
500
400
300
200
100
0
2536
4434
2952
2994
MAR
APR
MAY
JUN
JUL
AUG
SEPT
OCT
NOV
DEC
JAN
FEB
MAR
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SEPT
OCT
NOV
1976
1977

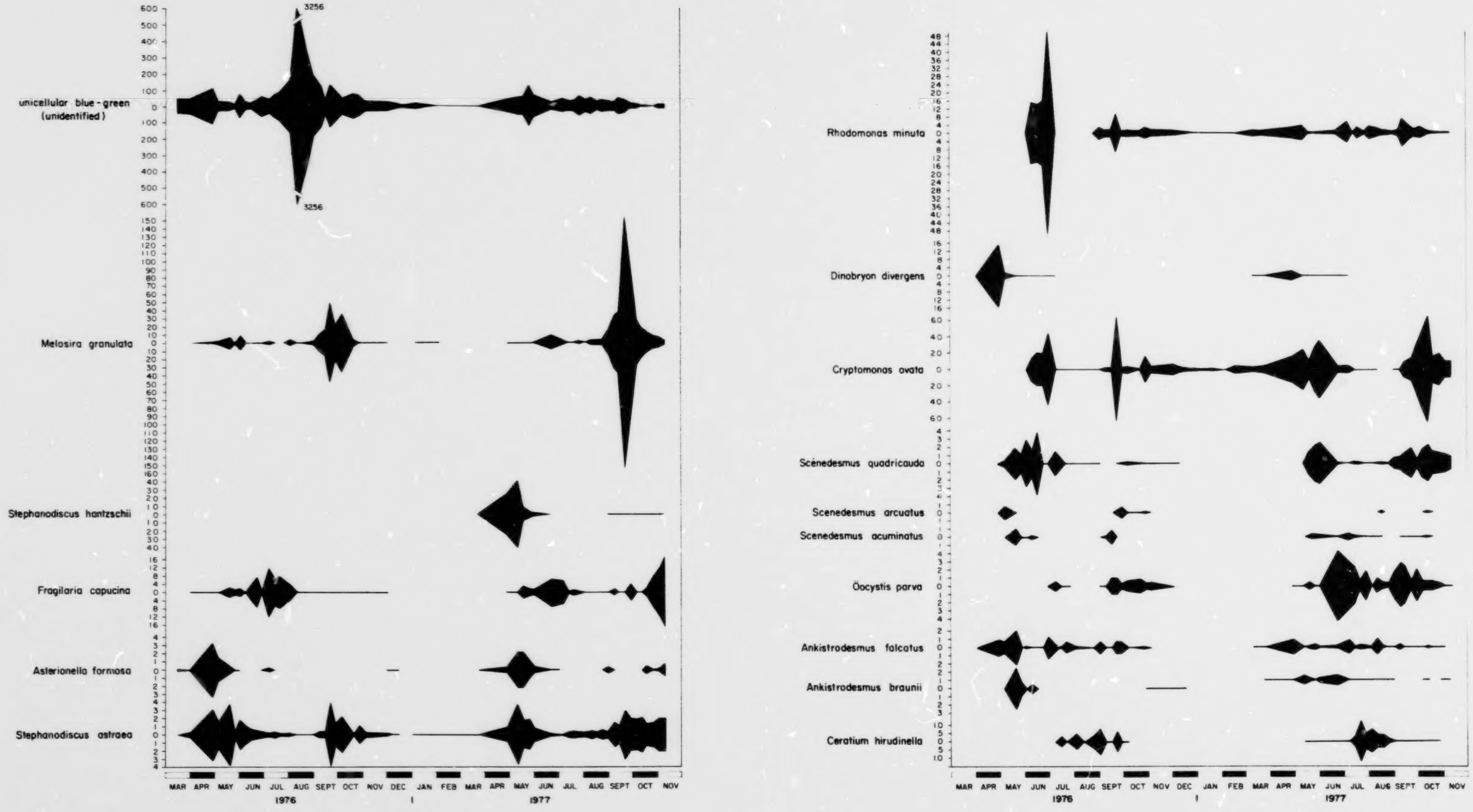

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FIG. 39 SEASONAL FLUCTUATIONS IN THE DOMINANT PHYTOPLANKTON SPECIES (cells x10^9 m^{-2}) IN BAPTISTE LAKE (NORTH)

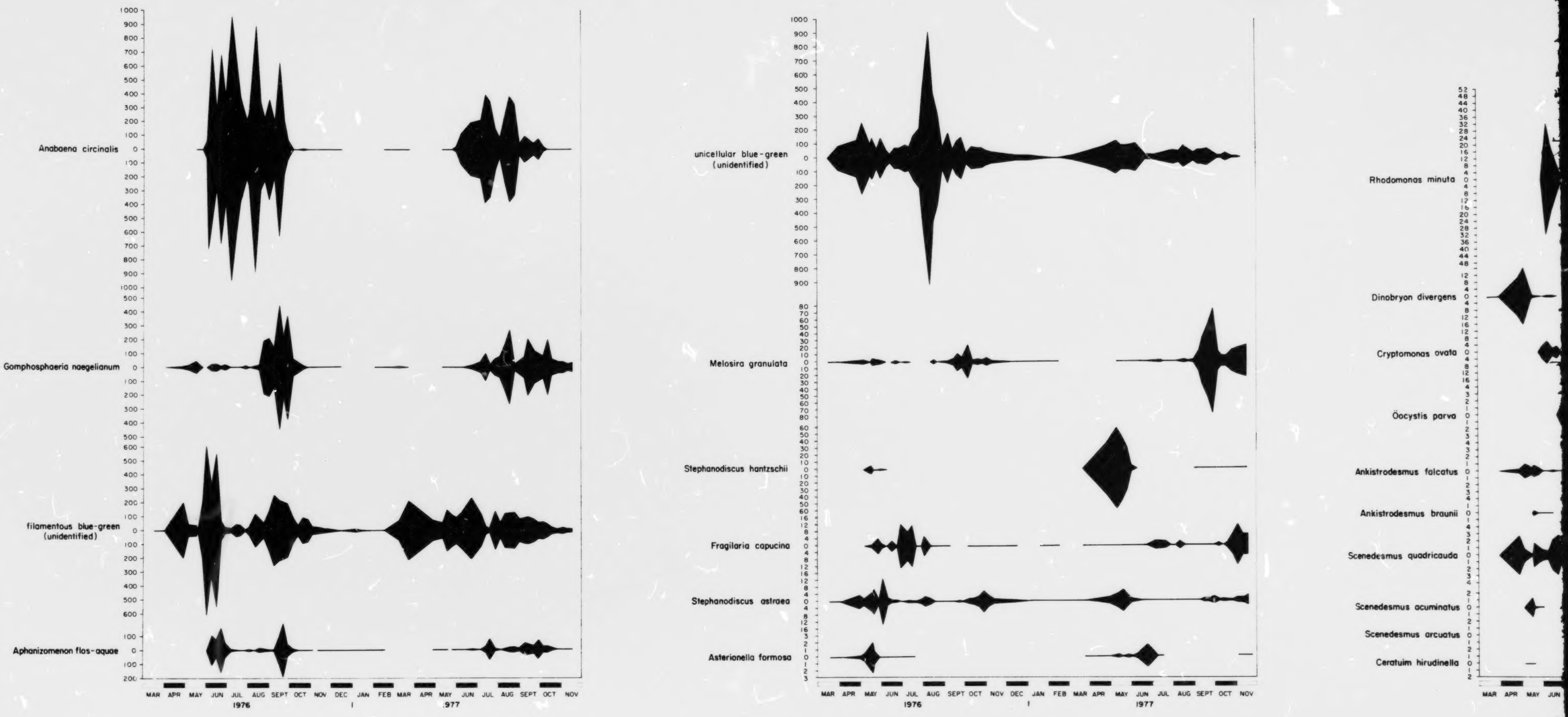

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FIG. 40 SEASONAL FLUCTU
(cells x 10^9 m^{-2}) IN E

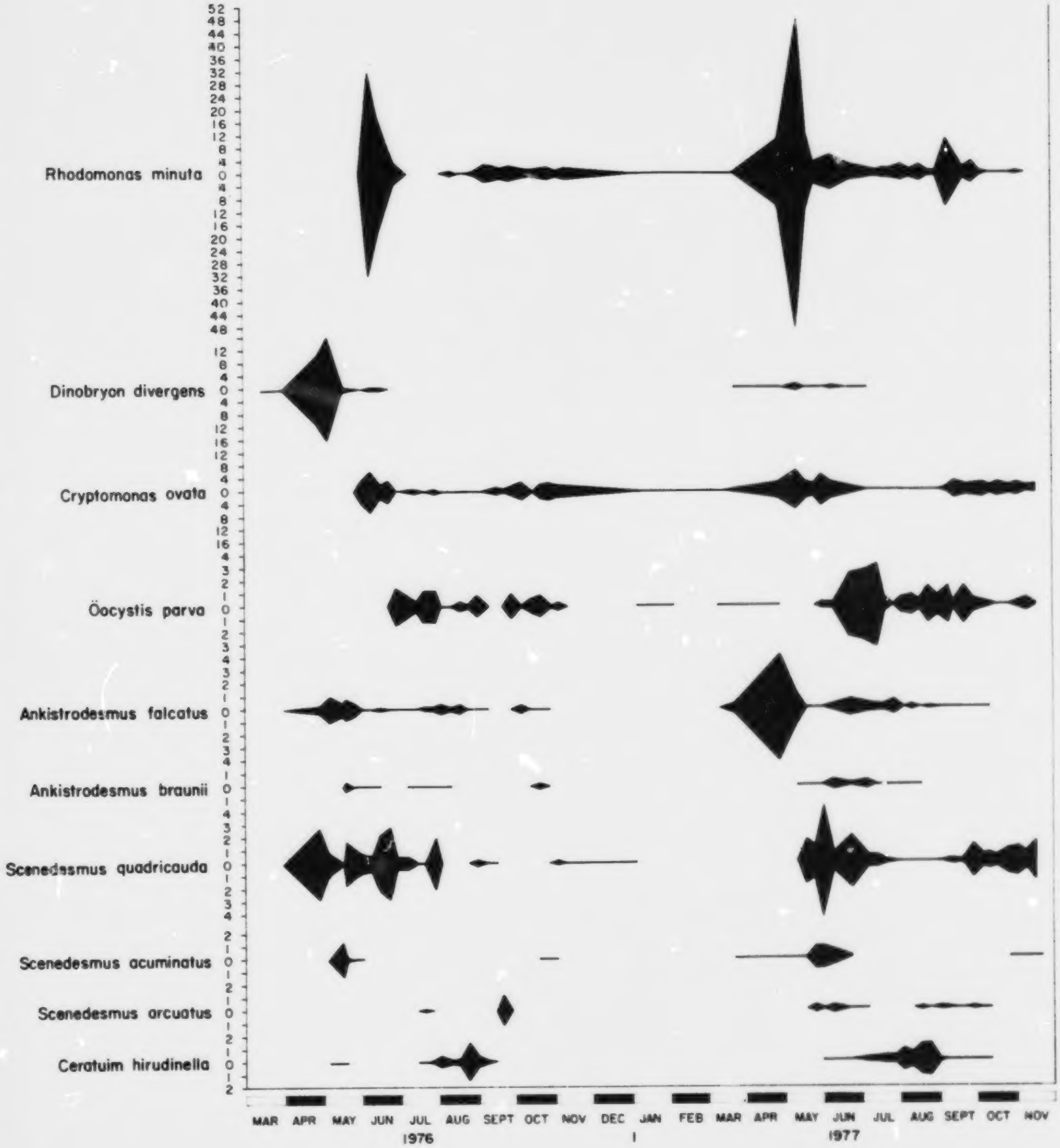

FIG. 40 SEASONAL FLUCTUATIONS IN THE DOMINANT PHYTOPLANKTON SPECIES (cells x 10^9 m^{-2}) IN BAPTISTE LAKE (SOUTH)

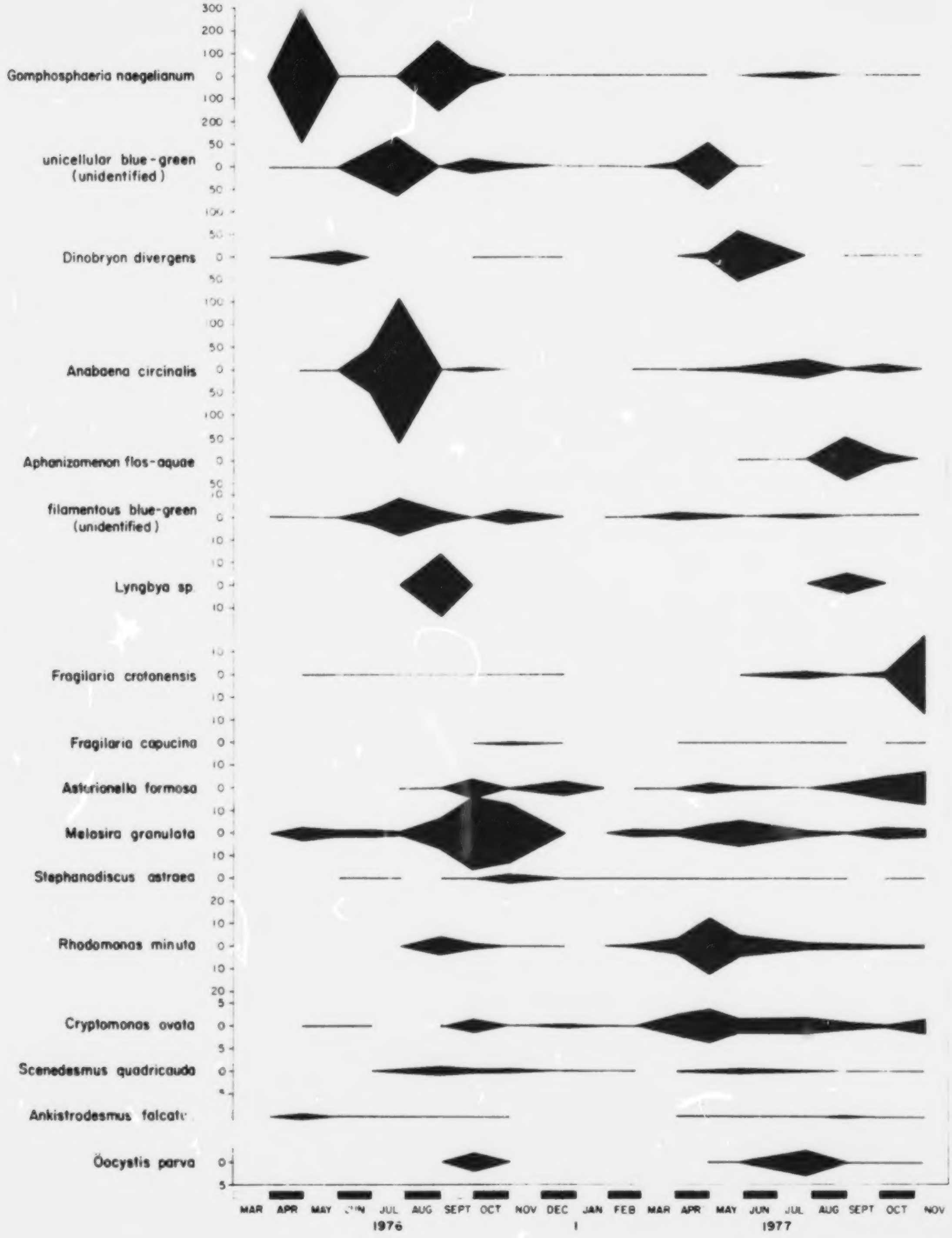

FIG. 41 SEASONAL FLUCTUATIONS IN THE DOMINANT PHYTOPLANKTON SPECIES (cells x 10^9 m^{-2}) IN GHOST LAKE

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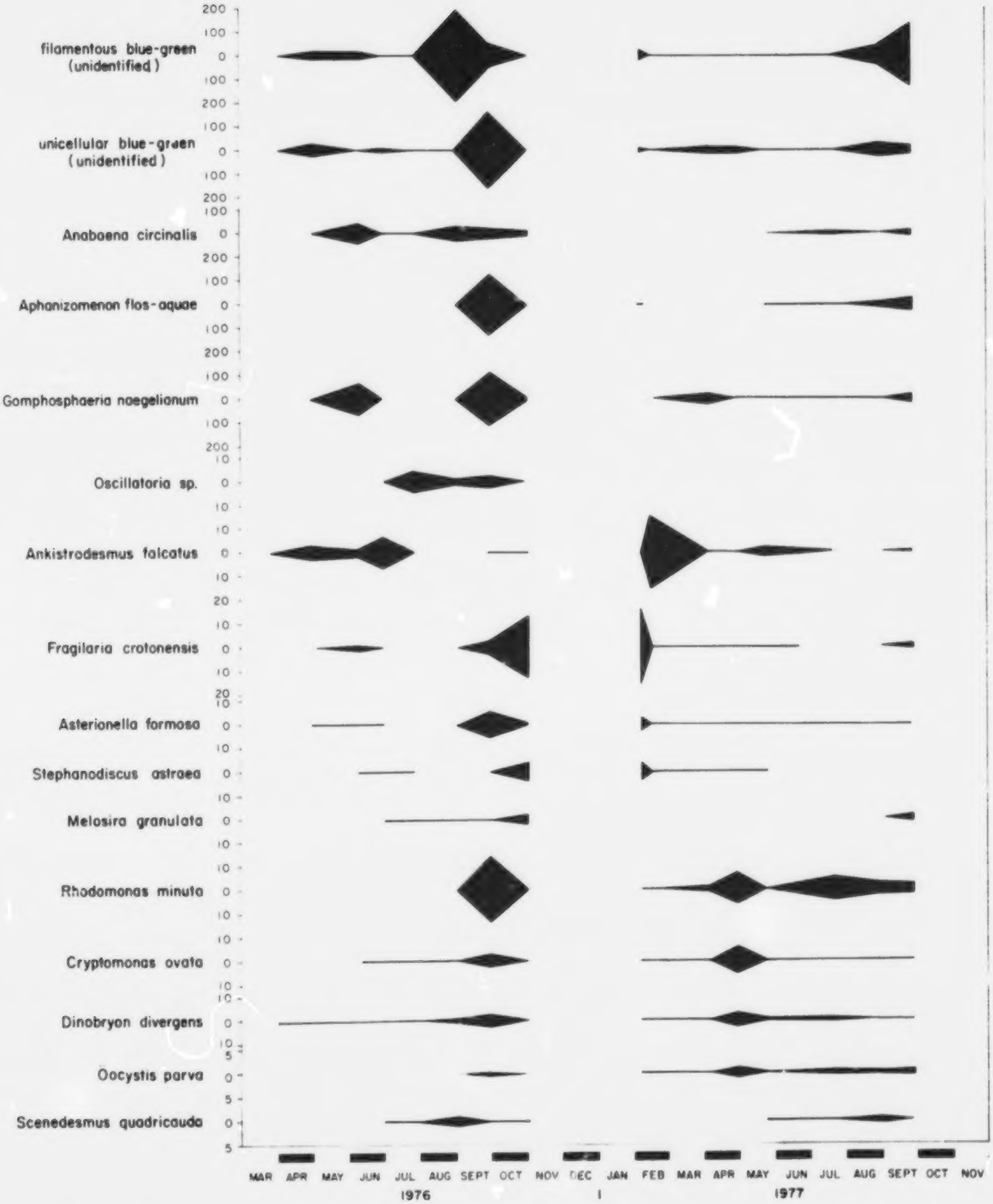

FIG. 42 SEASONAL FLUCTUATIONS IN THE DOMINANT PHYTOPLANKTON SPECIES (cells x 10^9 m^{-2}) IN ISLAND LAKE

Table 28. Phytoplankton cell count and cell volume open-water mean percent composition

	Baptiste (North)		Baptiste (South)		Ghost		Island	
	1976	1977	1976	1977	1976	1977	1976	1977
Cyanophyta								
Cell Number % Composition	97.9	91.3	97.7	90.9	90.0	56.9	95.0	90.3
Cell Volume % Composition	76.6	24.0	54.6	16.5	35.6	19.0	28.0	21.2
Chlorophyta								
Cell Number % Composition	0.2	1.6	0.2	1.4	0.6	3.2	0.7	4.1
Cell Volume % Composition	0.4	2.9	0.8	2.3	0.9	3.3	1.6	30.6
Bacillariophyta								
Cell Number % Composition	0.9	6.2	1.2	5.7	7.6	15.4	2.5	1.0
Cell Volume % Composition	10.4	56.0	22.0	48.5	51.7	33.3	63.1	7.1
Chrysophyta								
Cell Number % Composition	0.5	0.1	0.2	0.1	0.9	17.4	0.3	1.2
Cell Volume % Composition	1.5	0.2	1.4	0.3	3.9	21.8	2.4	7.1
Pyrrhophyta								
Cell Number % Composition	0.1	0.1	0.1	0.3	0.1	-	-	.1
Cell Volume % Composition	8.3	14.1	17.4	28.5	4.2	-	-	7.1
Cryptophyta								
Cell Number % Composition	0.5	0.8	0.6	1.8	0.9	8.4	1.4	3.4
Cell Number % Composition	2.7	2.9	3.5	3.9	3.7	12.6	4.8	27.0

of integral cell volume. This was because of the presence of fewer, but extremely large diatoms and dinoflagellates.

Many factors have been implicated in the seasonal succession of phytoplankton populations. These include light conditions, temperature, nutrient uptake kinetics, grazing, sinking, and parasitism (Reynolds 1980). A generalized explanation presented by Reynolds (1982) describes the periodic formation and collapse of near-surface density gradients as the major cause of phytoplankton periodicity in stratified lakes. This theory is based on the assumption that as a result of short-term changes in density stratification there will be a direct interaction with the morphological and behavioural characteristics of individual species, and indirect influences on the distribution of other physical and chemical factors influencing their survival.

The spring diatom assemblage in the north basin during 1976 was dominated by the large diatoms species Stephanodiscus astraea (3803 um^3) which was accompanied to a minor extent by Asterionella formosa (364 um^3) and Melosira granulata (335 um^3). Their combined areal cell volumes (cm^3 m^{-2}), or biomass, represented approximately 50 - 75% of total phytoplankton biomass during the month of May. Peridinium sp. (35,200 um^3) and an unidentified unicellular blue-green which may have been derived from broken colonies of Microcystis aeruginosa (48 um^3; R. S. Anderson, pers. comm.) were also signficant during May. Once thermal stratification became established in early June, the diatoms disappeared almost immediately although one species (Fragilaria capucina - 216 um^3) did persist in smaller numbers until early August.

The month of June represents a transition period between the extremes of vertical mixing and thermal stratification in Baptiste Lake, and as a consequence the phytoplankton community is characterized by short-term instability; i.e., several species make brief appearances. The crytophytes Rhodomonas minuta (105 um^3) and Cryptomonas ovata (1413 um^3) were present between early June and early July, and at peak concentrations represented approximately 10% of the biomass. The chryosphyte Synura sp. (113 um^3) was only detected on one occasion (June 15) but contributed to over 30% of the biomass on that date.

The cyanophytes were numerically the dominant group throughout the summer, and in the north basin their presence was evident from early June. The biomass was dominated by Anabaena circinalis (53 um^3) for most of June and July. In early August there was a shift to co-dominance by A. circinalis, Gomphospheria naegelianum (42 um^3), and the unidentified unicellular form. Another common cyanophyte in Alberta, Aphanizomenon flos-aquae (75 um^3), was present in June/July in relatively small amounts but disappeared in August.

The massive bloom of August 11/76 consisted of equal representation from all three species; this bloom coincided with the maximum surface temperature recorded that summer (22.0°C). The bloom event of August 30/76 consisted of A. circinalis and G. naegelianum in an approximate 2:1 ratio. The third bloom event (September 21/76) was again dominated by the latter two species and occurred during a late summer warming spell as the water temperature had risen from 14.5°C on September 15/76 to 17.0°C on the date of observation. The two latter blooms coincided with the development of thermal density instability in the water column at the end of August. The associated vertical mixing would have re-distributed some mass of hypolimnetic phosphorus and ammonia into the water column at that time.

The large, motile pyrrhophyte *Ceratium hirundinella* (67,196 um^3) was first observed on July 15 and represented a variable but important component, representing from 5 - 75% of the total biomass. It was most common at the beginning of August, but disappeared rapidly at the end of that month with the advent of isothermal conditions (15.0°C) in the water column.

Several other changes were observed in early September; the diatoms *Melosira granulata* and *S. astraea* commenced a rapid growth phase which also peaked on September 21/76. The blue-greens followed a similar growth phase during September after an early month depression; this phase included a reappearance of *A. flos-aquae*. Both groups declined through the remainder of the open-water season, assuming equal dominating importance to total biomass by mid-October.

The cryptophyte *C. ovata* also re-appeared late in October and was as important as the diatoms and blue-greens to total biomass immediately prior to freeze-up.

The winter minimum in the north basin was detected in mid-February, coinciding with the minimum in chlorophyll *a* concentrations and cell counts. In general, significant changes in the composition of the phytoplankton were slow to occur during the six months of ice-cover except near the end of the season.

Up until the winter minimum the community was dominated by two blue-greens (unidentified unicellular and G. naegelianum), the diatom S. astraea, and the cryptophyte C. ovata. The large pyrrhophyte Peridinium sp. ($35,200\ um^3$) appeared in late Feburary and was the largest single determinant of biomass up until the end of March. Two other species became important at this time: Mallomonas sp. ($2669\ um^3$) and Rhodomonas sp. The chrysophyte Dinobryon divergens ($625\ um^3$) also appeared in late March and was significant immediately after ice-out.

The phytoplankton community became significantly more diverse at ice-out. The blue-greens, diatoms, cryptophytes, and pyrrhophytes were all present in significant numbers.

The successional pattern during the open-water season of 1977 was similar to that observed during 1976, although the strengths of the various groups were different. In general, the total biomass of the diatoms was much higher during the spring/autumn pulses of 1977 than in the previous year. Conversely, the summer blue-green bloom was reduced relative to 1976.

These changes may have reflected significant differences in temperature and mixing regimes between years. The second year of study was cooler and the period of stratification was shorter and less intense. Stratification set in at a later date and disappeared earlier; furthermore, surface water temperatures in August were consistently lower in 1977.

The phytoplankton community was dominated by S. astraea (over 75%) for the entire month of May, although Peridinium sp. made another brief and significant appearance. The month of June was again a period of transition with all divisions except the chrysophytes being significantly represented. The chlorophytes were more prevalent in June of 1977 than in the previous year and were primarily represented by a Pandorina sp. (824 um^3) and Oocytis sp. (151 um^3).

S. astraea began rapid decline in the latter half of June but surprisingly persisted through the entire summer, representing a variable but significant component of the biomass (approximately 10%). In early June the pyrrhophyte C. hirundinella began a sustained growth phase which again carried through until the end of August; during this period it represented over 50% of the biomass. This species accompanied a bloom in the blue-greens which were dominated by A. circinalis throughout July and August. A shift to co-dominance by the unidentified "unicellular" and G. naegelianum occurred in late July. All forms declined simultaneously in August, unlike the previous year when we observed massive blooms during this month. One unique observation was that A. flos-aquae persisted throughout the entire summer of 1977 whereas it disappeared during August of 1976.

A massive autumn diatom pulse was initiated in early September of 1977 and was primarily dominated by S. astraea and M. granulata; these species dominated the biomass until freeze-up. The only other significant species during the autumn were Pandorina sp., C. ovata and G. naegelianum.

The spring bloom in the south basin in 1976 was dominated in terms of biomass by a diatom assemblage consisting primarily of Stephanodiscus astraea; Melosira granulata, Asterionella formosa, and Peridinium sp. were also present in smaller quantities. Immediately after ice-out the cyanophytes and chlorophytes were also significant. The diatoms, representing over 50% of the biomass during the latter half of May, dominated through until June 8, 1976 after which point the blue-greens rapidly became established. The diatoms flourished during a period of cool temperatures (maximum surface temperature 14.5°C), and extensive vertical mixing. The ratio of mixing depth of euphotic zone depth decreased from 4.1 (April 28) to 1.6 (June 8) during this pulse.

Two cryptophytes, Rhodomonas minuta and Crytomonas ovata, made a significant contribution (approximately 15%) to the biomass estimates during the June transition from diatom to blue-green domination.

The cyanophytes generally dominated the biomass (over 50%) from early June to mid-September, although there was a shift from domination by Anabaena circinalis and the unidentified unicellular form in late August to co-domination with Gomphosphaeria naegelianum, Aphanizomenon flos-aquae, and an unidentified filamentous form at the end of that month. The latter shift was coincident with increasing phaeophytin concentrations in the epilimnion, decreasing water temperatures, and a 2 m suppression in the thermocline. The latter process may also have introduced hypolimnetic nutrients into the euphotic zone.

The large, motile pyrrhophyte Ceratium hirundinella was also present in variable numbers from June to September and was extremely important in terms of total cell volumes, even though the blue-greens dominated in terms of cell counts. During a "bloom" of this species recorded on August 24, 1976, it represented 72% of the areal phytoplankton biomass.

By the last week in September and during the entire month of October, the phytoplankton was again dominated by the diatoms Melosira granulata and Stephanodiscus astraea, but some cryptophytes were also present. Temperatures were below 15°C after September 27, and the ratio of mixing depth to euphotic depth was greater than 3.1 after this period.

During the period of ice-cover, the phytoplankton was dominated by five species: the large diatom S. astraea, the unidentified unicellular blue-green, the cryptophyte C. ovata, and two species which appeared in March, the pyrrhophyte Peridinium sp., and the chrysophyte Mallomonas sp.

The pattern of community change in the south basin during the open-water season 1977 was similar to that of 1976, although the abundances of the various divisions were again different. In general, the diatoms were present in much greater numbers during spring and autumn, and in summer the blue-greens were present in diminished numbers.

The south basin underwent a complete spring mixing during 1977, and the strong showing by the diatoms may have reflected an increased silica supply as suggested by changes in the hypolimnetic silica concentrations (Figure 33).

On May 25, 1977, the diatoms represented 91% of the areal cell volume and this was almost entirely due to S. astraea. During June the community was diverse with all divisions being represented. In early July the community was dominated by the blue-greens and dinoflagellates in roughly equal proportions, although later in July and all through August the dinoflagellates represented over 60% of the biomass. Once again, this biomass was primarily determined by Ceratium hirundinella.

This species disappeared rapidly in early September following the same pattern as observed during 1976. This was correlated with declining water temperatures (< 15.0°C) and an increase in the mixing depth. A shift in the composition of the cyanophyte community similar to that which occurred in 1976 was also observed at this time, the primary change being the reduction of the dominating Anabaena circinalis and a moderate increase in Aphanizomenon flos-aquae. The remaining three cyanophytes were ubiquitous from late July to through late September.

Other than a single large cryptophyte pulse (Cryptomonas ovata) which occurred on September 28, 1977, the phytoplankton biomass in September and October was dominated by the diatoms Melosira granulata and Stephanodiscus astraea. A large hypolimnetic pool of silica was mixed into the upper water column during this period (Figure 33) and may have contributed to their success.

The general successional pathways for both basins for both years are presented in Table 29. The patterns are similar to those described by Reynolds (1982) for eutrophic lakes, which are presented in the same table. The gross changes in the phytoplankton community can be at least partially correlated with mixing processes in the water column.

The phytoplankton samples from Ghost and Island Lakes were taken at monthly intervals and, therefore, only a cursory overview

Table 29. Generalized phytoplankton successional pathways for eutrophic lakes (Reynolds 1982) and Baptiste Lake.

Baptiste Lake

Dominants	May	June	July
	Stephanodiscus astraea Peridinium spp. Microcystis aeruginosa	Stephanodiscus astraea Anabaena circinalis Aphanizomenon flos-aquae	Anabaena circinalis Microcystis aeruginosa Ceratium hirundinella
Sub-dominants	Cryptomonas ovata Gomphosphaeria naegelianum	Cryptomonas ovata Microcystis aeruginosa	Stephanodiscus astraea Gomphosphaeria negelianum

Eutrophic Lakes

Spring

Asterionella formosa
Fragilaria crotonensis
Stephanodiscus astraea
Stephanodiscus hantzschii
Ankistrodesmus spp.

Eudor spp.
Pandorina spp.
Volvox spp.
Ankyura judayi

Table 29. continued

August	September	October
Anabaena circinalis Microcystis aeruginosa Ceratium hirundinella Gomphosphaeria naegelianum	Anabaena circinalis Microcystis aeruginosa Aphanizomenon flos-aquae Gomphosphaeria naegelianum Stephanodiscus astraea Melosira granulata	Stephanodiscus astraea Melosira granulata Cryptomonas ovata
Stephanodiscus astraea	Crytomonas ovata	Gomphosphaeria naegelianum Microcystis aeruginosa

Summer	Autumn	
Anabaena spp. Aphanizomenon flos-aquae Gloeotrichia spp.	Ceratium hirundinella Microcystis aeruginosa Gomphosphaeria circinalis	Asterionella formosa Stephanodiscus astraea Oscillatoria spp.
Asterionella formosa Melorisa granulata Closterium aciculare	Fragilaria crotonensis Staurastrum pingue	

of phytoplankton periodicity in those lakes can be presented. As is evident from the Baptiste Lake data, several species or divisions may be important for very short periods of time and, therefore, the full range of fluctuations may not be well represented by monthly data.

The vertical profiles of temperature and oxygen for Ghost Lake suggest that it did not mix thoroughly during the spring of 1976. The diatoms were unimportant in the spring bloom of that year, perhaps reflecting the lack of the appropriate circulation. Over 90% of the total phytoplankton biomass was dominated by the blue-green G. naegelianum at ice-out. Similarily, at the end of May, the community was singley dominated by the motile chrysophyte Dinobryon divergens. As observed in Baptiste Lake, the community in June was significantly more diverse, but primarily dominated by the blue-greens. The most important species was A. circinalis; sub-dominants included the unidentified unicellular blue-green, G. naegelianum, and three diatoms (F. crotonensis, M. granulata, and S. astraea).

Anabaena circinalis continued to be important through to the end of July, and at this time the pyrrhophyte C. hirundinella was observed in significant quantities. By the end of August, a shift in the blue-green community was evident with the dominance of a large, unidentified species of Lyngbya. This species was short-lived and before the end of September the phytoplankton community was completely dominated by the diatoms Asterionella formosa, S. astraea, and M. granulata. The highest biomass of the year was observed just prior to freeze-up in Ghost Lake; this consisted almost entirely of a large bloom of S. astraea although some C. ovata was also encountered, paralleling the composition in Bapiste Lake.

The latter two species displayed similar proportions in late December, with the large cryptophyte C. ovata disappearing in mid-winter but returning to dominate the community in March; S. astraea was significant through until March. The community became more diverse in late winter; the cryptophyte Rhodomonas minuta, the pyrrohphyte Peridinium sp., and the diatom M. granulata were all important in March.

The lake appeared to mix thoroughly in the spring of 1977, and a more significant diatom pulse was recorded than in the previous year; the order of dominance was S. astraea, M. granulata, and A. formosa. Dinobryon divergens was equally important at ice-out and as in the previous year dominated the community at the end of May. No sample was taken in June 1977, but by the end of July the community was dominated by the blue-green A. circinalis and a smaller, unidentified Anabaena sp.

The pattern for the remainder of the year was similar to that of 1976. Lyngbya sp. dominated in late August, leading into an autumn pulse of diatoms and C. ovata. One significant difference was that the diatom pulse was dominated by A. formosa instead of S. astraea in October and by F. crotonensis in November.

The phytoplankton community in the north basin of Island Lake exhibited unique features compared to either community in Ghost or Baptiste Lakes. The spring pulse was not dominated by the diatoms in 1976 or 1977, perhaps reflecting the lack of vertical circulation in this partially meromictic basin. Certain chlorophyte species not significant in the other lakes were important in Island. A plate of blue-green algae was occasionally encountered at 10.0 m., and in September of both years a plate of purple sulphur bacteria was found at this depth. During September this plate was within the thermocline.

Immediately after ice-out in 1976 the community was dominated by Ankistrodesmus falcatus (241 u^3) and D. divergens. The biomass was low at this time but by early June a significant development of blue-greens had occurred. The dominant species were the same as those observed in the other two lakes: A. circinalis and G. naegelianum. The only significant diatom species in either May or June was F. crotonensis.

An unusually depauperate community was encountered in late July. At this time the biomass was at the lowest point of open-water season; the community was temporarily dominated by an unidentified Oscillatoria sp., although several other blue-green species were present. In early September the community was still dominated by the blue-greens, but by the end of the month the community had diversified. The blue-greens were still dominant, but the cryptophytes had also become important following the trend at Baptiste Lake. Also Aphanizomenon flos-aquae made an important contribution to total biomass at this time.

Isothermal conditions were established by early November, the water column was free to circulate, and a large diatom pulse was observed. The highest phytoplankton biomass of the open-water season was observed on this date, and the community was almost entirely dominated by S. astraea; F. crotonensis was also present.

An unusually rich diatom community was observed in early February in Island Lake, at a time when Ghost and Baptiste were at their annual biomass minima. The two species that dominated declined rapidly after this point as revealed by a second February sample, which was dominated by the chlorophytes A. falcatus and Peridinium sp. In early April the community diversified again; at ice-out all divisions were represented with C. ovata and D. divergens dominating, as observed in 1976.

In late May two chlorophyte species not previously significant were important components of the biomass: Pandorina sp and Closterium acerosum. According to the scheme presented by Reynolds (1982), these species are characteristic of the algal assemblages of eutrophic systems (Table 29).

No samples were taken in June, but in late July the community was dominated by a mixed assemblage of blue-greens and cryptophytes. The large pyrrhophyte C. hirundinella made a single, but important appearance at this time. The minimum biomass for the open-water season of 1977 was recorded in late August; A. flos-aquae was the most important species. The last sample for the year was taken in late September; at this time the community was dominated by the blue-greens and cryptophytes following the late summer trend in the other lakes.

7.8 PHYTOPLANKTON PRIMARY PRODUCTION

The *in situ* measurement of phytoplankton carbon fixation in Baptiste Lake was conducted between October 20, 1976 and November 16, 1977 using the radiocarbon method of Steemann Nielsen (1952). Experiments were carried out at the principal sampling stations in each basin, although the incubations were not simultaneous because of the sampling and travel time taken between stations. In general, the north basin incubation was initiated approximately one hour after the start of the south basin incubation, which generally ran from 1100 to 1500 hours.

During the period of study, 33 experiments were conducted in the south basin and 29 in the north. The experiments were carried out at approximately weekly intervals during the open-water seasons and fortnightly intervals during the period of ice cover.

The discrete mean hourly rates of photosynthesis (mg C m^{-3} hr^{-1}) have been integrated for the 0-10 m (BN) and 0-20 m (BS) intervals of the water column. These hourly integrals (mg C m^{-3} hr^{-1}) have been transformed to daytime integrals (mg C m^{-2} day^{-1}) by applying an experimentally-derived diurnal factor to the results of the short-term experiments. The annual estimates of carbon fixation were derived by integrating the daytime rates over the 12 month period. Mean daytime rates were calculated by dividing annual rates by 365; these rates are illustrated in Figure 43..

The derivation of daytime production estimates from short-term incubations have been discussed in detail by Anderson (1974(b)). Three basic approaches have been employed, including

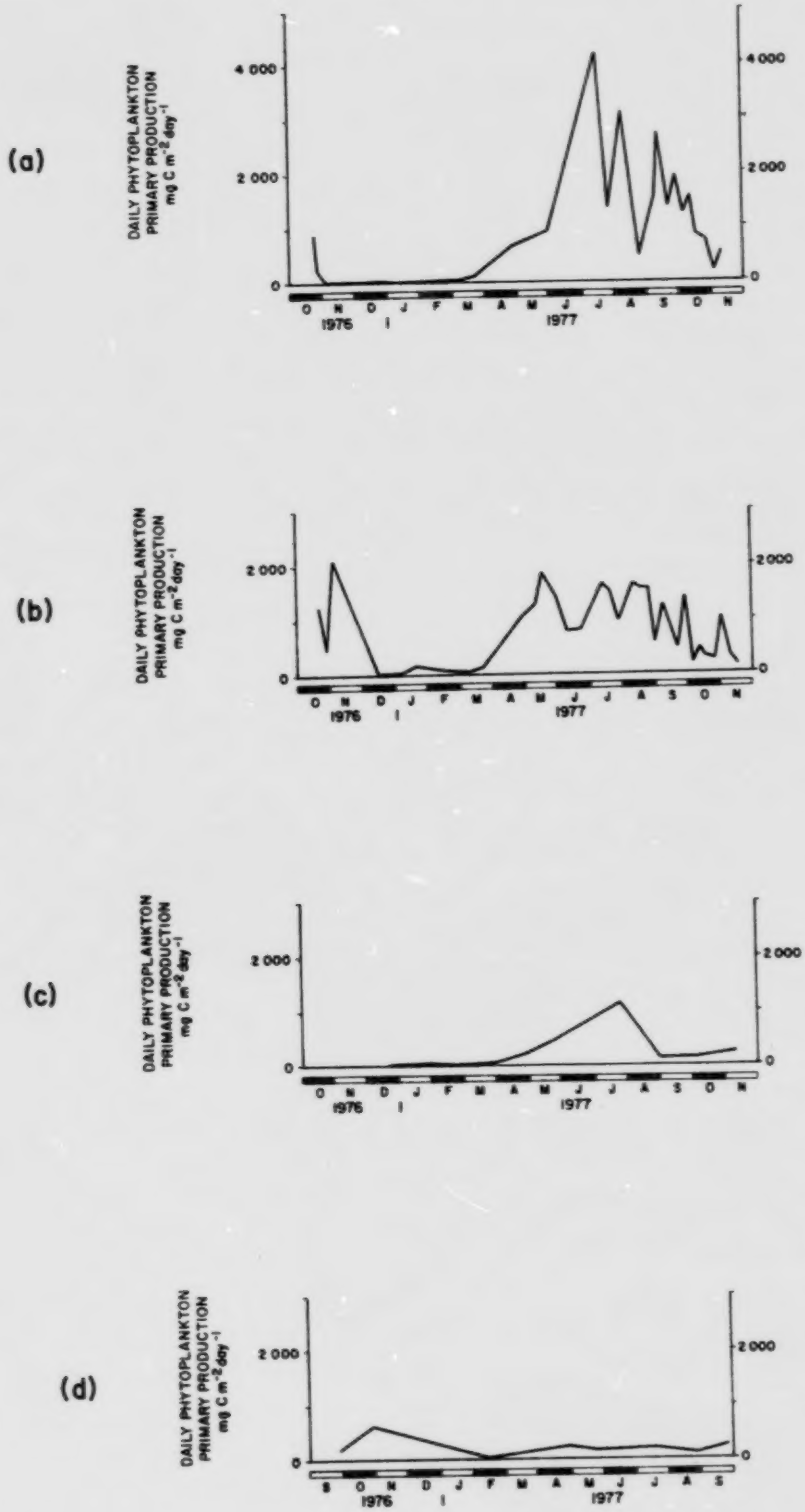

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FIG.43 SEASONAL FLUCTUATIONS IN PHYTOPLANKTON PRIMARY PRODUCTION FOR (a) BAPTISTE LAKE (NORTH), (a) BAPTISTE LAKE (SOUTH), (c) GHOST LAKE AND (d) ISLAND LAKE

extrapolations from (a) the fraction of a given lakes's daily production occurring in a defined time interval (b) the fraction of daily irradiance occurring during the incubation interval and (c) the fraction of daylight hours occurring during the incubation interval. However, diurnal variations in photosynthetic rates can introduce large errors into simple extrapolations such as in (b) and (c) above. Furthermore, waterbodies of different morpho-edaphic characteristics may display distinctly different diurnal patterns. According to Hammer et al (1973), no simple universal extrapolation technique exists for the calculation of daily production.

Accordingly, we carried out three diurnal studies on Baptiste Lake to provide a data base for calculating such an extrapolation factor applicable to that lake alone.

The incubations were conducted during spring (May 30-31, 1977), summer (July 18-19, 1977), and autumn (October 11-12, 1977). On each day, four successive incubations were carried out ranging in duration from approximately 3.0 to 4.5 hours. During the spring and summer studies an overnight incubation was also carried out.

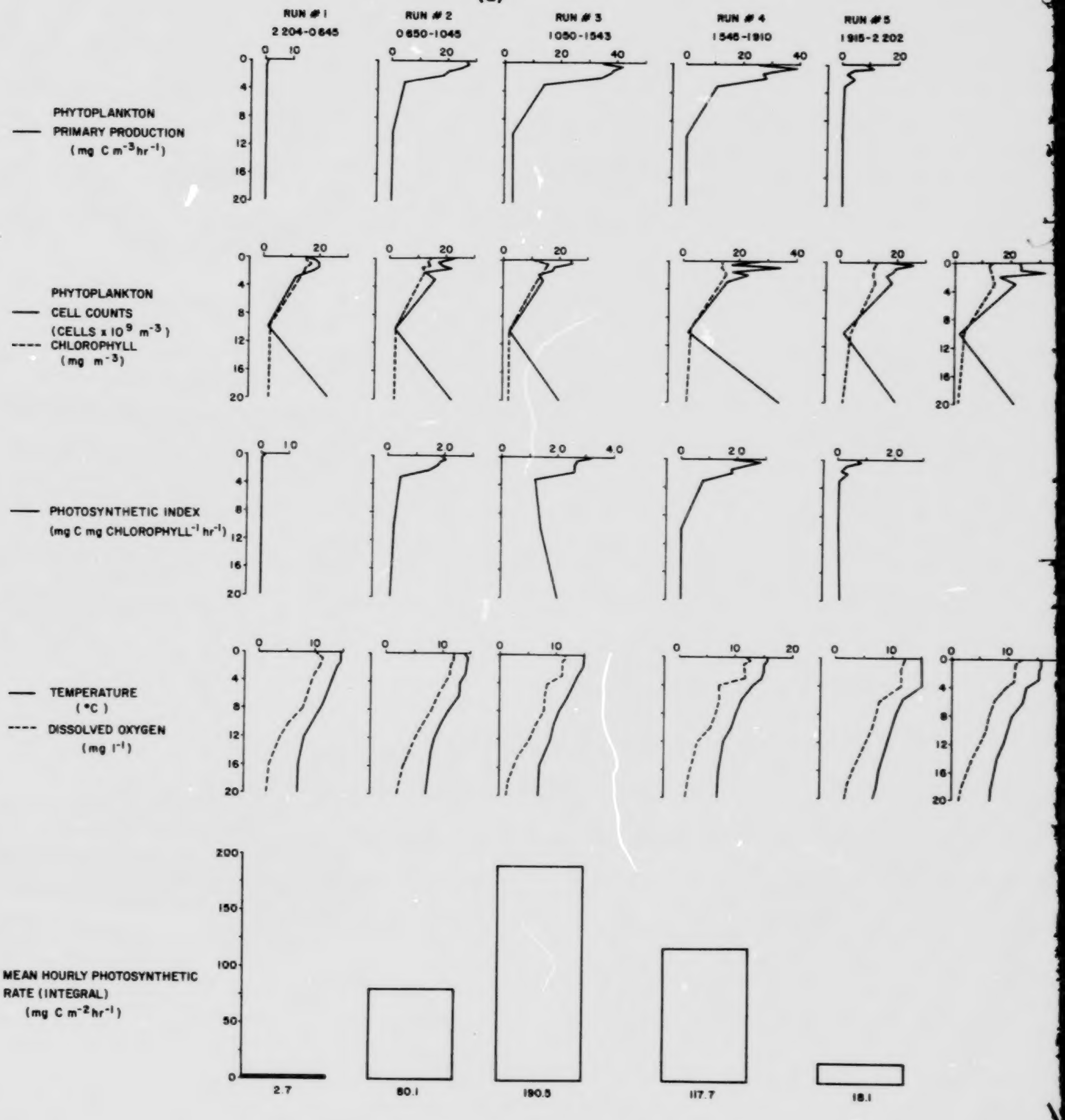
Integral photosynthesis was determined for each incubation period and the cumulative rate then determined for the 24 hour period. The overnight integrals were negligible in relation to the daily integral, and thus the lack of such a run on Oct. 11-12/77 is not viewed as a significant omission. The data for each diurnal study are illustrated on Figure 44.

During the May 30-31 study, the maximum hourly volumetric and integral rates of photosynthesis occurred during the #3 run (1050 to 1545). Surface inhibition was observed on all incubations except the overnight run; the depth of the maximum rate of photosynthesis (A_{max}) was consistently observed at 0.5 m during the day. Chlorophyll a concentrations displayed no significant diurnal cycle and thus the photosynthetic index (mg C mg chlorophyll^{-1} hr^{-1}) displayed a maximum value during the #3 run. Irradiance data were unavailable for this experiment. However, the cloud cover was approximately 50% for most of the day.

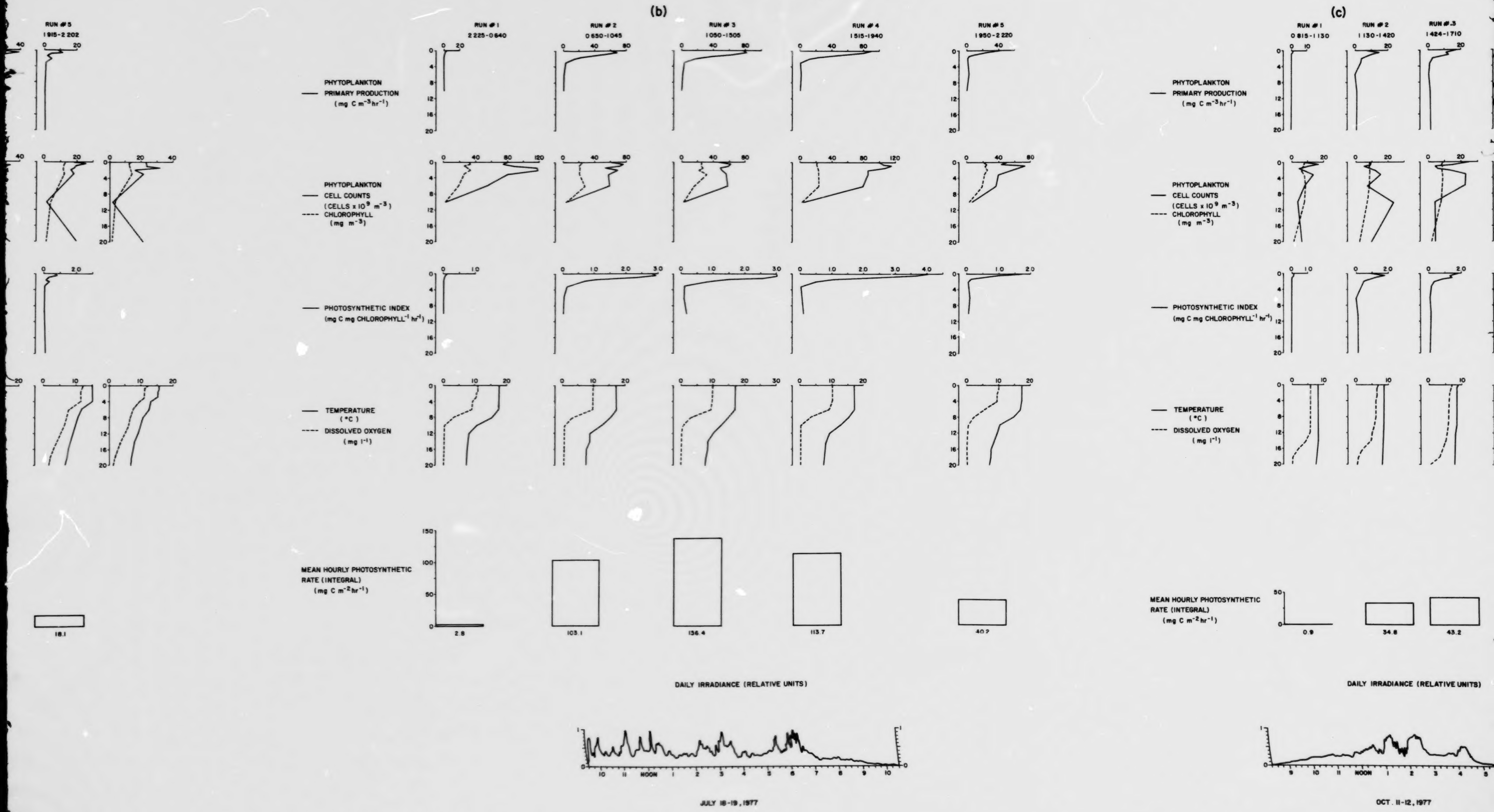
During the July 18-19 study, the maximum hourly integral rate was measured during approximately the same time interval as in the May 30-31 study, i.e. 1050 - 1505 hours. The maximum volumetric rates for all depths except 0 m were also measured during this incubation. However, the maximum hourly rate at the surface was observed in the 1515 to 1940 hours interval. The depth of A_{max} during runs #1 and #5 was also at 0 m.

A somewhat different diurnal pattern emerged from the Oct. 11-12 study in that an afternoon peak was observed (run #3, 1425 - 1710 hours), and A_{max} was found at 0 m. The maximum volumetric rates for the day for the 1.0 and 1.5 m depths were also found during this run. However, the integral for run #3 was undoubtedly influenced by ^{14}C - uptake at 10.0 and 20.0 m. When integrated through a ten meter water column even the comparatively low rates of 1.1 mg C m^{-3} hr^{-1} at 10.0 m and 1.8 mg C m^{-3} hr^{-1} at 20.0 m can influence the integral rate of unproductive days significantly. The cause of ^{14}C - uptake at such depths where light levels are generally below the 1% level may be attributable to dark fixation by bacteria (Wetzel 1975).

(a)



MAY 30-31, 1977


(b)
(c)
RUN # 5
1915-2202
RUN # 1
2225-0640
RUN # 2
0630-1045
RUN # 3
1050-1505
RUN # 4
1515-1940
RUN # 5
1950-2220
RUN # 1
0815-1130
RUN # 2
1130-1420
RUN # 3
1424-1710
PHYTOPLANKTON
PRIMARY PRODUCTION
(mg C m^{-3} hr^{-1})
PHYTOPLANKTON
CELL COUNTS
(CELLS x 10^9 m^{-3})
CHLOROPHYLL
(mg m^{-3})
PHOTOSYNTHETIC INDEX
(mg C mg CHLOROPHYLL^{-1} hr^{-1})
TEMPERATURE
(°C)
DISSOLVED OXYGEN
(mg l^{-1})
MEAN HOURLY PHOTOSYNTHETIC
RATE (INTEGRAL)
(mg C m^{-2} hr^{-1})
0
4
8
12
16
20
0
20
40
80
120
1.0
2.0
3.0
4.0
10
30
50
100
150
18.1
2.8
103.1
136.4
113.7
40.2
0.9
34.8
43.2
DAILY IRRADIANCE (RELATIVE UNITS)
1
0
9
10
11
NOON
1
2
3
4
5
6
7
8
9
10
JULY 18-19, 1977
OCT. 11-12, 1977

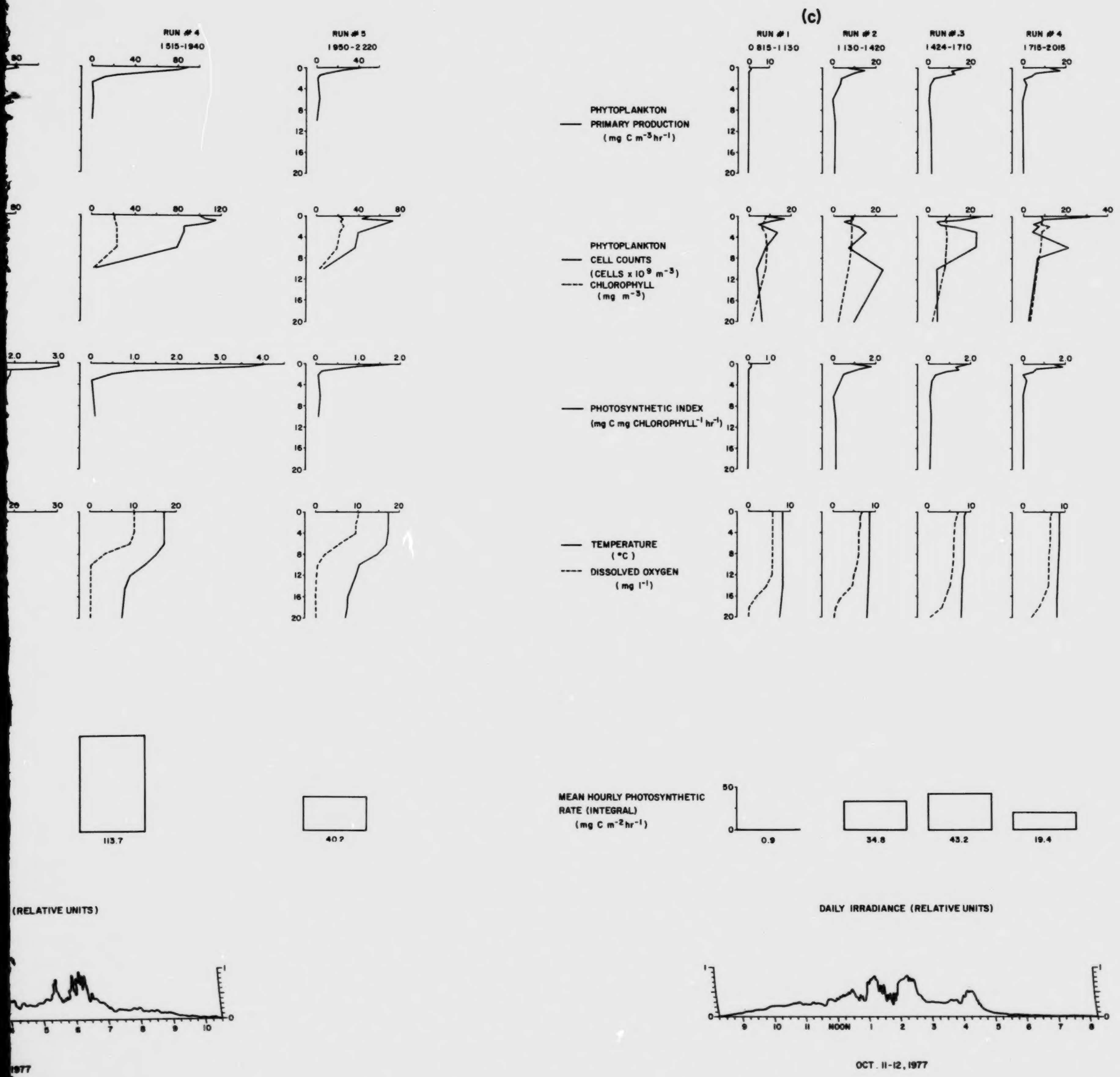


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FIG. 44 VERTICAL DISTRIBUTION OF SELECTED PARAMETERS DURING THE DIURNAL STUDIES OF (a) MAY 30-31, 1977, (b) JULY 18-19, 1977, AND (c) OCTOBER 11-12, 1977 IN BAPTISTE LAKE

Because of the myriad of taxonomic, nutritional, and physical variables which could influence photosynthetic activity on any given day, we have decided not to apply these diurnal data too rigorously, i.e., as being representative of the individual seasons in which they were generated. Instead, we have combined the data to produce a single set of extrapolation factors which were employed for the year. It is acknowledged that this approach may not account for all possible variations in diurnal activity that may be encountered. However, without an extensive data base for this phenomenon it is difficult to justify any other approach.

The mean hourly production rate for each incubation interval was expressed as a percentage of the daytime cumulative production rate. Each daytime hour during each of the three days was assigned such a percentage value. The mean hourly percentage was then determined for the three days to provide a gross indication of the proportion of daily production occurring each hour.

The results are presented in Table 30 for the daytime hours of 1000 to 1700 hours. The data indicate that during the 1000 to 1400 hours interval approximately 33% of daytime production occurs. This figure is close to the figure of 35% generated by Noton (1974) for Lake Wabamun during summer months.

The daytime rates of integral photosynthesis for all basins are illustrated in Figure 43. The discrete hourly rates are summarized in the Appendix in Tables A14 to A17. The hourly integrals, daytime integrals and integral chlorophyll a are presented in Tables 31 and 32. The annual daytime production in the north basin of Baptiste was estimated at 355.3 g C m^{-2} and

Table 30. Percentage of total daytime photosynthesis occurring each hour.

Date	1000-1100	1100-1200	1200-1300	1300-1400	1400-1500	1500-1600	1600-1700
May 30 - 31	5.6	-	10.6	10.6	10.6	10.0	8.3
July 18 - 19	6.9	8.5	8.5	8.5	8.5	7.2	7.1
Oct. 11 - 12	0.3	6.5	12.7	12.7	14.6	15.7	15.7
x %	4.3	7.5	10.6	10.6	11.2	11.0	10.4

Table 31. Mean hourly integral photosynthesis (ΣA_H; mg C m^{-2} hr^{-1}) for 0-3.0 m and 0-10.0 m, daytime integral photosynthesis (ΣA_D; mg C m^{-2} day^{-1}), and integral chlorophyll a concentration (ΣB; mg chl a m^{-2}) at Baptiste North.

Date	0-3.0 m ΣA_H	0-10.0 m ΣA_H	%	ΣA_D	ΣB
Oct 20/76	56.6	92.3	61.3	847.0	240.4
Oct 27	12.1	23.3	51.9	212.6	156.3
Nov 03	24.8	75.5	32.8	703.9	101.4
Dec 15	0.8	0.8	100.0	7.3	5.5
Dec 22	0.7	2.2	31.8	20.2	4.2
Jan 05/77	1.0	1.1	90.9	10.4	3.9
Jan 19	1.1	2.3	47.8	21.6	1.9
Feb 02	1.4	1.4	100.0	12.4	2.4
Feb 16	2.5	2.6	96.2	24.2	3.6
Mar 09	5.1	5.8	87.9	53.9	4.9
Mar 23	5.1	11.0	46.4	105.9	15.5
Apr 27	49.7	71.8	69.2	677.5	111.7
May 31	57.3	101.4	56.5	936.7	93.7
Jul 18	399.6	424.4	94.2	4275.4	447.1
Jul 27	140.3	148.8	94.3	1362.6	275.5
Aug 10	288.0	311.3	92.5	3149.0	545.2
Aug 17	186.0	196.1	94.9	1869.8	400.2
Aug 24	41.9	52.1	80.4	482.8	222.9
Aug 31	101.7	110.6	92.0	1033.8	417.1
Sep 08	147.7	155.6	94.9	1509.4	376.9
Sep 13	238.2	271.4	87.8	2734.7	568.5
Sep 21	120.8	140.5	86.0	1316.7	447.5
Sep 28	124.5	161.9	76.9	1490.3	356.4
Oct 05	116.2	134.3	86.5	1228.3	361.5
Oct 12	120.1	135.9	88.4	1592.2	271.9
Oct 17	86.7	93.9	92.3	864.5	204.4
Oct 26	56.8	83.8	67.8	772.0	161.6
Nov 02	11.0	19.5	56.4	179.6	150.2
Nov 09	34.5	61.4	56.2	563.3	149.4

Table 32. Mean hourly integral photosynthesis (ΣA_H; mg C m^{-2} hr^{-1}) for 0-3.0 m and 0-20.0 m, daytime integral photosynthesis (ΣA_D; mg C m^{-2} day^{-1}), and integral chlorophyll a concentration (ΣB; mg chl a m^{-2}) at Baptiste South.

Date	0-0.3 m ΣA_H	0-20.0 m ΣA_H	%	ΣA_D	ΣB
Oct 20/76	104.8	133.9	78.3	1261.7	352.6
Oct 27	34.6	47.9	72.2	442.3	467.5
Nov 03	60.5	225.3	26.9	2108.3	348.9
Nov 10	12.5	18.1	69.1	167.0	158.3
Dec 15	1.9	1.9	100.0	17.2	14.0
Jan 05/77	1.5	1.5	100.0	13.1	7.7
Jan 19	1.7	18.2	9.3	164.3	4.8
Feb 16	0.9	6.6	13.6	64.5	4.2
Mar 09	3.8	5.2	73.1	49.5	5.4
Mar 23	13.8	13.8	100.0	127.8	26.8
Apr 28	69.6	101.6	68.5	1063.1	191.2
May 11	104.7	137.6	76.1	1285.5	186.7
May 18	127.3	192.6	66.1	1869.4	200.4
May 31	100.7	151.0	66.7	1409.2	121.0
Jun 08	56.2	82.8	67.9	789.7	134.0
Jul 13	167.5	179.0	93.6	1660.6	296.5
Jul 19	126.6	150.0	84.4	1536.0	211.3
Jul 27	104.4	104.4	100.0	960.7	226.6
Aug 10	174.6	174.6	100.0	1670.9	355.4
Aug 17	173.7	173.7	100.0	1585.7	726.7
Aug 24	36.3	175.1	20.7	1591.6	636.2
Aug 31	56.7	61.1	92.8	558.5	218.5
Sep 08	132.7	140.0	94.8	1295.7	304.6
Sep 21	26.9	47.7	56.4	444.3	229.2
Sep 28	96.6	155.5	62.1	1413.9	434.1
Oct 05	19.5	20.5	95.1	187.5	129.2
Oct 12	17.9	49.6	36.1	488.7	132.2
Oct 17	22.3	49.3	45.2	320.7	116.0
Oct 26	26.5	28.0	94.6	256.9	111.5
Nov 02	72.2	108.4	66.6	1058.1	127.5
Nov 09	17.5	35.0	50.0	317.8	128.9
Nov 16	12.8	18.2	70.3	167.4	135.2

the south basin at 249.6 g C m^{-2} for the period November 3, 1976 to November 2, 1977. The higher annual production rate in the north basin was reflected in substantially higher daytime rates on most occasions as compared to the south basin. However, the intensity of sampling was not as frequent as that in the south basin due to the inclement weather conditions on several occasions. As a result, the integration in the north basin has been performed on fewer data points. These annual rates result in the lake being classified as eutrophic as defined by the criteria of Likens et al. (1977).

In order to determine any basin to basin differences in integral photosynthesis, the data have been tested statistically using the Wilcoxon signed-rank matched-pairs test. This is a non-parametric test for the comparison of median values (Siegel 1956). Differences were considered significant at the 0.10 probability level. Data were paired for the 24 dates from Tables 31 and 32 on which data for both basins were available (all seasons). We found a significant difference between the mean hourly integrals ΣA_H in the 0-3.0 m interval (2 tailed $P = 0.092$) but not in the daytime integrals ΣA_D (2 tailed $P = 0.391$) which were calculated through 0-10.0m (BN) and 0-20.0 m (BS) intervals.

The mean daytime rates of photosynthesis have been estimated at 973.4 mg C m^{-2} day^{-1} for Baptiste north and 683.8 mg C m^{-2} day^{-1} for Baptiste south. The total content of organic carbon (TOC) in the water column of Baptiste ranges from approximately 300 to 500 gm C m^{-2} (unpublished data) and thus the daytime phytoplankton photosynthetic production of organic carbon represents much less than 1% of the total pool.

Photosynthesis exhibited marked fluctuations over the annual cycle, concomitant with the changes in phytoplankton biomass and chlorophyll a concentrations reported earlier. The daytime integrals at Baptiste North ranged from less than 20 mg C m^{-2} day^{-1} in the December-February period, to the maximum rate of 4275.4 mg C m^{-2} day^{-1} recorded on July 18/77. During late February and early March there was an increase in primary production under the ice as light penetration and the photoperiod increased.

The peak production period extended from mid-July to mid-September, with daytime integrals usually in excess of 1300 mg C m^{-2} day^{-1}. During spring and late autumn the daytime productivity at this site was depressed to the 500 - 800 mg C m^{-2} day^{-1} range; similar rates were also encountered during the brief open-water period of 1976 which was included in this experimentation (October 20 - November 3, 1976).

Both hourly (ΣA_H) and daytime (ΣA_D) integral photosynthesis were strongly correlated with integral chlorophyll a at Baptiste North (winter date included).

$$\mathrm{Log}_{10}\ \Sigma A_H = 1.016\ \mathrm{Log}_{10}\ \Sigma B - 0.550\ (n = 29,\ r^2 = 0.85)$$
$$\mathrm{Log}_{10}\ \Sigma A_D = 1.041\ \mathrm{Log}_{10}\ \Sigma B + 0.192\ (n = 29,\ r^2 = 0.60)$$

The seasonal pattern at Baptiste South was slightly different from Baptiste North, reflecting the between basin differences in integral chlorophyll a concentrations described previously. Spring and autumn productivity was more intense in the south basin than in the north basin and the summer production was comparatively lower; no integrals in excess of 2000 mg C m^{-2} day^{-1} were recorded in the south basin although three were recorded in the north basin.

A more detailed picture of the spring bloom and early summer minimum is available at Baptiste South for reasons discussed above. The spring peak in integral photosynthesis occurred on May 18, 1977 when a value of 1869.4 mg C m^{-2} day^{-1} was recorded; this had risen from a production rate of 1063.1 mg C m^{-2} day^{-1} immediately after ice-out on April 28, 1977. During the transition period in June the productivity was low, in the 800 mg C m^{-2} day^{-1} range. By early July the production had quickly returned to the 1500 mg C m^{-2} day^{-1} level but on no occasion did measurements exceed the peak value recorded on May 18, 1977. During October and November the production was generally less than 500 mg C m^{-2} day^{-1}. However, unusually productive events were recorded on almost identical dates of 1976 (November 3) and 1977 (November 2). This was extremely late in the season with cool, isothermal conditions prevailing. Both events were characterized by high concentrations of the diatom Stephanodiscus astraea, however, as was the peak event on May 18, 1977. These events may be reflective of significant seasonal taxonomic variations in photosynthetic capacity.

Significant correlations were also found between hourly/daytime integral photosynthesis and integral chlorophyll a concentrations at Baptiste South:

$$\mathrm{Log}_{10}\ \Sigma A_H = 0.791\ \mathrm{Log}_{10}\Sigma B + 0.067 \quad (n = 33,\ r^2 = 0.68)$$

$$\mathrm{Log}_{10}\ \Sigma A_D = 0.794\ \mathrm{Log}_{10}\Sigma B + 1.028 \quad (n = 33,\ r^2 = 0.67)$$

These regression analyses indicate that daytime production in Baptiste Lake can be estimated from a knowledge of chlorophyll a concentration alone.

In order to characterize the vertical distribution of photosynthetic activity, the discrete, mean, hourly carbon fixation rates have been plotted on depth profiles (Figures 46 to 49). Unlike chlorophyll a concentrations, phytoplankton photosynthesis displayed marked stratification in the upper water column, with maximum rates (A_{max}) usually observed near the 0.5 m depth. During the open-water season of 1977, surface inhibition of photosynthesis was observed on most dates at both Baptiste south (18 out of 23 experiments) and north (11 out of 18 experiments). Accordingly, the photosynthetic maxima were more frequently observed at the surface (0m) of Baptiste north (7 out of 18 experiments) as opposed to Baptiste south (5 out of 23 experiments). One obvious pattern in the depth of A_{max} was elucidated: it was observed at 0 m in both basins on the dates of surface chlorophyll a maxima (Aug. 10 and 17, 1977).

The mean, hourly maximum rates of photosynthesis (A_{max}) were highly correlated with the corresponding chlorophyll a concentrations (B) in both basins during the open-water season.

Baptiste North

$$\mathrm{Log}_{10}\ A_{max} = 1.339\ \mathrm{Log}_{10}\ [B] - 0.290 \quad (n = 21,\ r^2 = 0.82)$$

Baptiste South

$$\mathrm{Log}_{10}\ A_{max} = 0.753\ \mathrm{Log}_{10}\ [B] + 0.632 \quad (n = 27,\ r^2 = 0.64)$$

In order to provide a quantitative illustration of the depth pattern of carbon fixation, integral photosynthesis was calculated for the 0-3.0 m interval and then compared with the total water column integrals (0-10.0 m at BN and 0-20.0 m at BS). The data are presented in Tables 31 and 32, with the comparisons being expressed as a percentage of total ΣA_H found in the top three meters. Note that the 0-3.0 m interval roughly correponds to the calculated euphotic zone depths presented for the preceding year in Tables 22 and 23, and represents the zone in which the phytoplankton was usually homogenously distributed. Unfortunately, light extinction data were not obtained regularly during 1977 due to equipment failure and thus corresponding z_{eu} data could not be generated. However, this 0-3.0 m zone in fact corresponds closely to the major zone of photosynthetic activity, as borne out by the data.

Based on a paired date comparison of 18 observations from the open-water interval in both basins, we observed that approximately 75.7% of ΣA_H was in the 0-3.0 m interval at Baptiste north and 69.9% at Baptiste south. During the July-August period of maximum photosynthesis, well over 90% of the activity was concentrated in this interval. Thus, as the standing crop increases, the euphotic zone diminishes and the higher rate of photosynthetic activity is progressively compressed into a shallower band. Conversely, during the spring and autumn the percentage of ΣA_H occurring in the top three meters was greatly reduced, corresponding to increased water circulation/light transmission and decreased phytoplankton standing crop.

Recent analyses completed during the OECD Eutrophication Study (1981) have revealed strong relationships between annual integral photosynthesis (ΣA) and both annual total phosphorus concentration [TP] and annual euphotic zone chlorophyll a concentration [B], i.e.:

$$\Sigma A = 31.1\ [TP]^{.54} \quad (n = 49,\ r = 0.71)$$

$$\Sigma A = 56.6\ [B]^{.64} \quad (n = 49,\ r = 0.79)$$

In order to place the annual productivity estimates from Baptiste Lake in a broader perspective, the above empirical relationships have been used to predict ΣA for Baptiste Lake. The mean annual [TP] and [B] were 0.070 mg L^{-1} and 22.5 mg m^{-3} respectively. The measured mean annual rate of carbon fixation in Baptiste Lake (if calculated as the mean of the individual basin rates) was approximately 302.5 gm C m^{-2} yr^{-1} during the study period. The predicted ΣA are 308.4 gm C m^{-2} yr^{-1} (TP) and 378.1 gm C m^{-2} yr^{-1} (B) respectively. Thus, it would appear that the productivity and nutrient regime of Baptiste Lake is not unlike those of the other north temperate lakes used to formulate the OECD empirical relationships. The observed data fall within the 95% confidence intervals of the OECD relationships (Figures 6.9 and 6.9, OECD 1981).

One of the major confounding factors of in situ primary productivity measurements is the day to day and within day variability in irradiance. Photosynthetic activity can be either enhanced or suppressed by ambient light conditions in relation to the standing crop present. If sufficient measurements coincide with either extreme of the light regime, then high or low biases will be introduced into the annual production estimates. Clearly, this is a problem which can only be overcome by taking a large number of measurements or by conducting experiments in a suitable light environment representative of long-term irradiance trends (Fee 1978).

Thus, the fewer observations available for in situ photosynthesis, the larger the potential for high or low values to bias the annual rate estimates. This is the case for the production estimates from Ghost and Island Lakes. The measurements of in situ photosynthesis could only be conducted at monthly intervals due to logistic constraints, and thus the annual production rates should be interpreted with caution. At best, the data provide some indication of the depth profiles of photosynthesis in less productive, nearby lakes (Figures 50 and 51).

The annual productivity rates for Ghost and Island Lakes have been estimated at 99.3 gm C m^{-2} yr^{-1} and 81.4 gm C m^{-2} yr^{-1} respectively. The average daytime production rates were estimated at 277.7 mg C m^{-2} day^{-1} and 251.4 mg C m^{-2} yr^{-1}.

7.9 Particulate Organic Seston

Determinations of ash-free dry weight or particulate organic seston were made on each sampling trip. Theoretically, this parameter is non-specific to the primary producers since any particulate organic material retained by the 80 um Wisconsin net could contribute to the observed concentrations. Alternatively, many small phytoplankters may pass through the mesh and thus a negative bias may be introduced into the estimates. Based on the range of cell volumes encountered during the analysis of preserved phytoplankton samples (taken with a Van Dorn bottle) from this study, it would now appear that the vertical haul technique has introduced a serious error into the particulate organic seston data base. Not surprisingly, we found very poor statistical correlations between this parameter and any of the phytoplankton indices (photosynthetic rate, chlorophyll a, cell volume, and cell numbers) when expressed on a comparative basis. A further complication to such comparisons is the fact that the seston samples were retrieved continuously through the water column, whereas the phytoplankton indices are integrations of discrete samples, which may not have accounted for phenomena such as hypolimnetic algal plates (Fee 1980).

The data for Baptiste north, Baptiste south, Ghost and Island are illustrated in Figure 45. Significant seasonal cycles are evident, generally following the patterns of the phytoplankton indices. The mean open-water data are summarized in Table 33 for each basin. The open-water concentrations were higher in all basins during 1976 than in 1977, except for Island Lake.

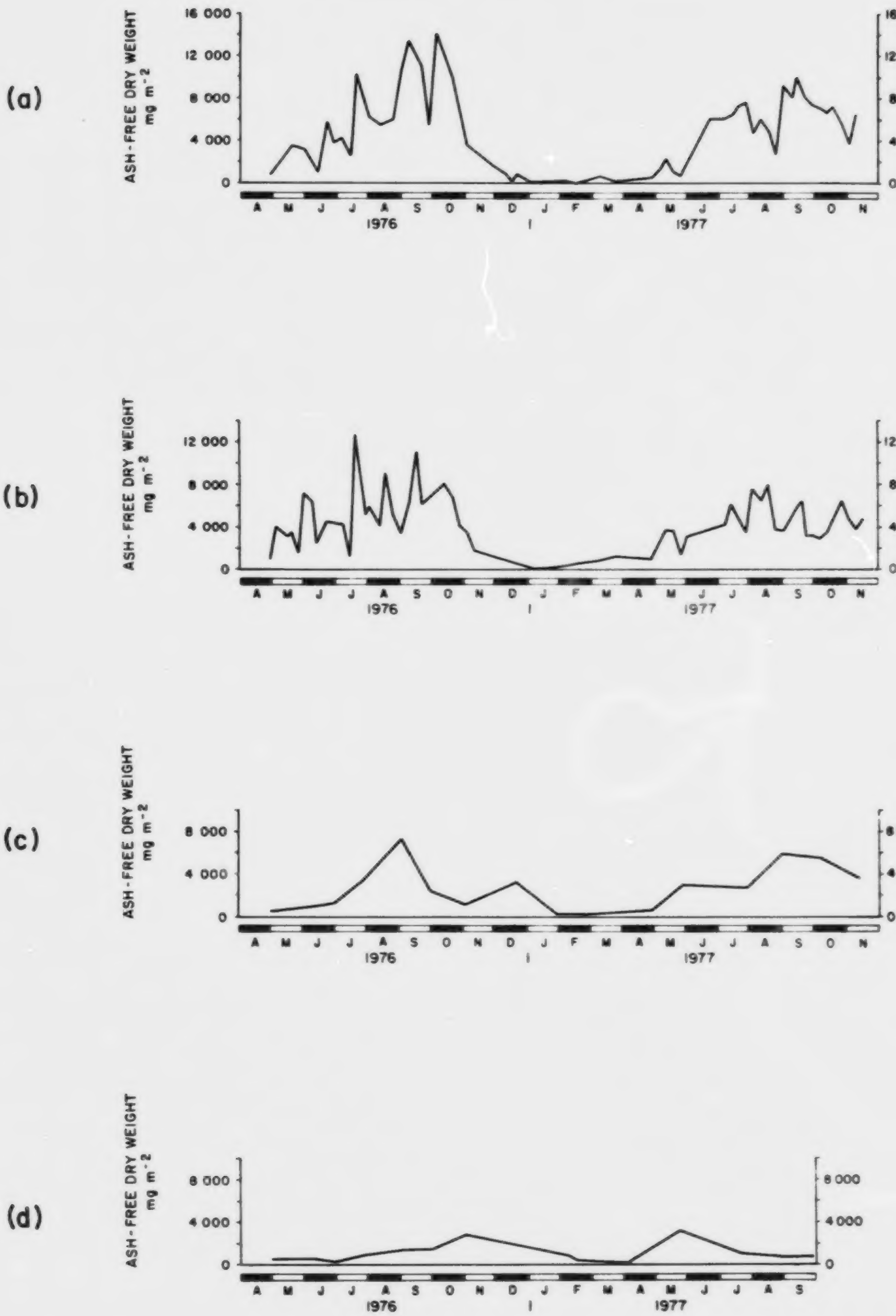

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FIG. 45 SEASONAL FLUCTUATIONS IN ASH-FREE DRY WEIGHT VALUES FOR (a) BAPTISTE LAKE (NORTH), (b) BAPTISTE LAKE (SOUTH), (c) GHOST LAKE AND (d) ISLAND LAKE

Table 33. Mean ash-free dry weight concentrations (mg m^{-2}).

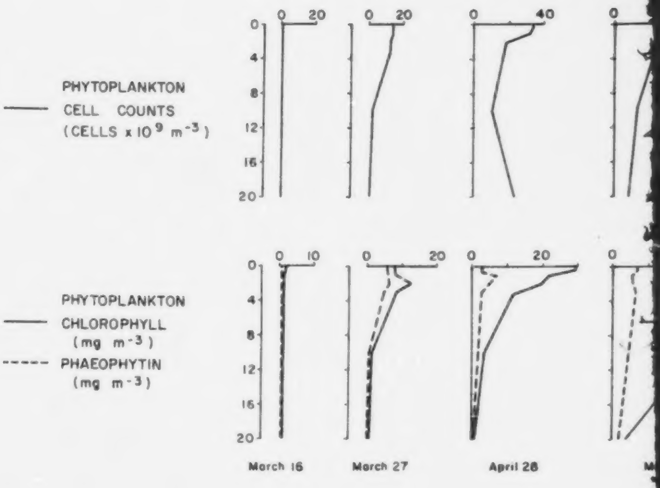
		Baptiste North	Baptiste South	Ghost	Island
1976	open-water season	6747	5226	3826	1119
1977	open-water season	5623	4432	3601	1457

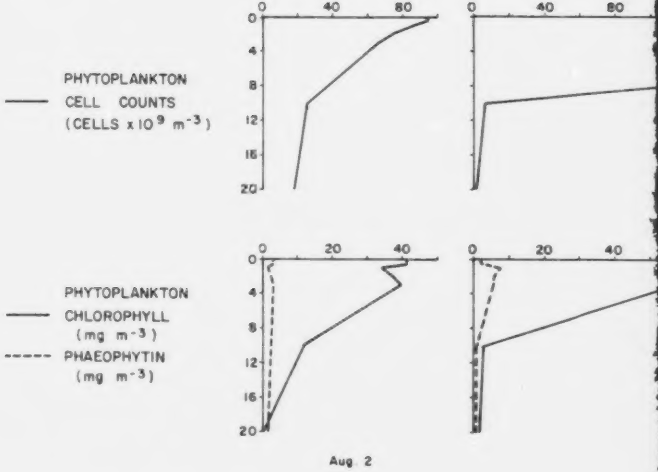
7.10 BAPTISTE LAKE ZOOPLANKTON

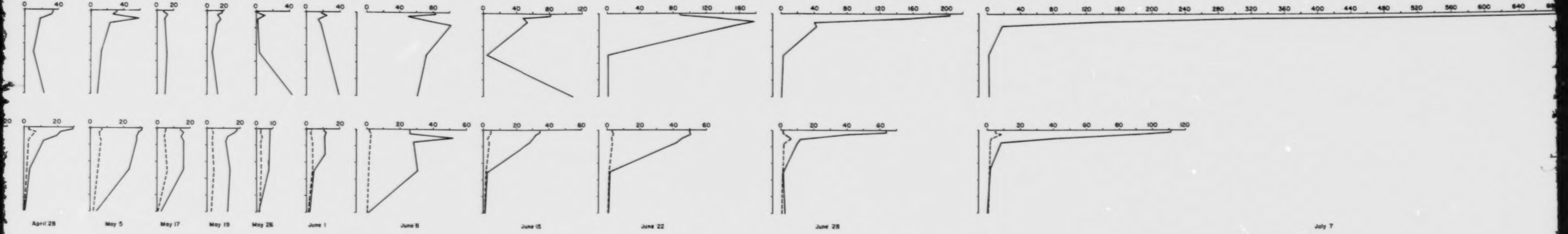
In Baptiste Lake, as in most North American lakes, the zooplankton comprises two major groups of small invertebrates: the Crustacea, primarily copepods and cladocerans, and the Rotatoria, the rotifers. Only those animals large enough to be retained in an 80 um mesh plankton net are described here; this includes most zooplankton found in Alberta lakes. The complete list of species identified can be found in the Appendix in Table A22 with their relative dominance for Baptiste Lake summarized in Tables A23 to A24. This analysis and summary was prepared by P. Mitchell.

7.10.1 COPEPODA

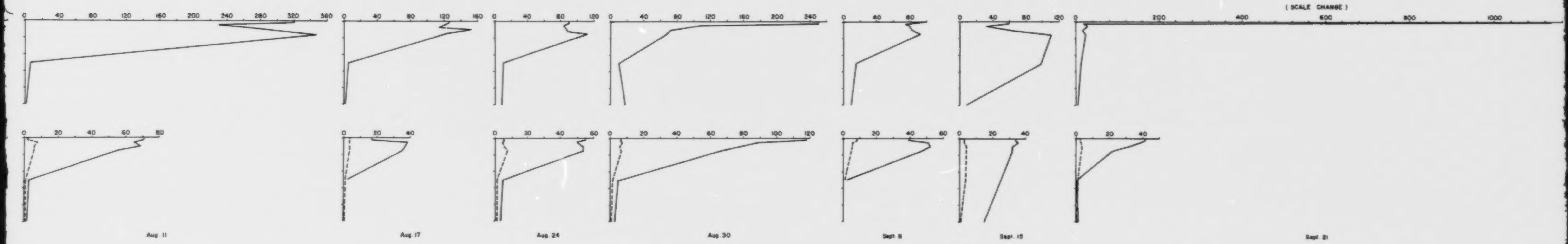
The dominant copepod in the two basins of Baptiste Lake, and in Island Lake, was the cyclopoid Diacyclops bicuspidatus thomasi. This species is one of the most common zooplankters in north-central Alberta lakes, and was a dominant species present in the St. Lawerence Great Lakes (Watson and Carpenter 1974; Patalas 1972). It was not observed in samples from Ghost Lake; although it may have been present in small numbers. Diacyclops was very abundant in early spring in Baptiste and Island Lakes, at a time when populations of other species were relatively low. Many of these individuals were breeding adults, as was also observed by Chymko (1977) in Lesser Slave Lake in early spring. Populations of Diacyclops remained high until mid-July both years in the deeper southern basin of Baptiste, but had declined by the end of June in the northern basin. Only a few individuals were observed in August samples. Diacyclops is known to enter a diapause period during the time


PHYTOPLANKTON
CELL COUNTS
(CELLS x 10⁹ m⁻³)
PHYTOPLANKTON
CHLOROPHYLL
(mg m⁻³)
PHAEOPHYTIN
(mg m⁻³)
0
20
40
10
0
4
8
12
16
20
March 16
March 27
April 28


PHYTOPLANKTON
CELL COUNTS
(CELLS x 10⁹ m⁻³)
PHYTOPLANKTON
CHLOROPHYLL
(mg m⁻³)
PHAEOPHYTIN
(mg m⁻³)
0
40
80
20
0
4
8
12
16
20
Aug. 2



1976



1976

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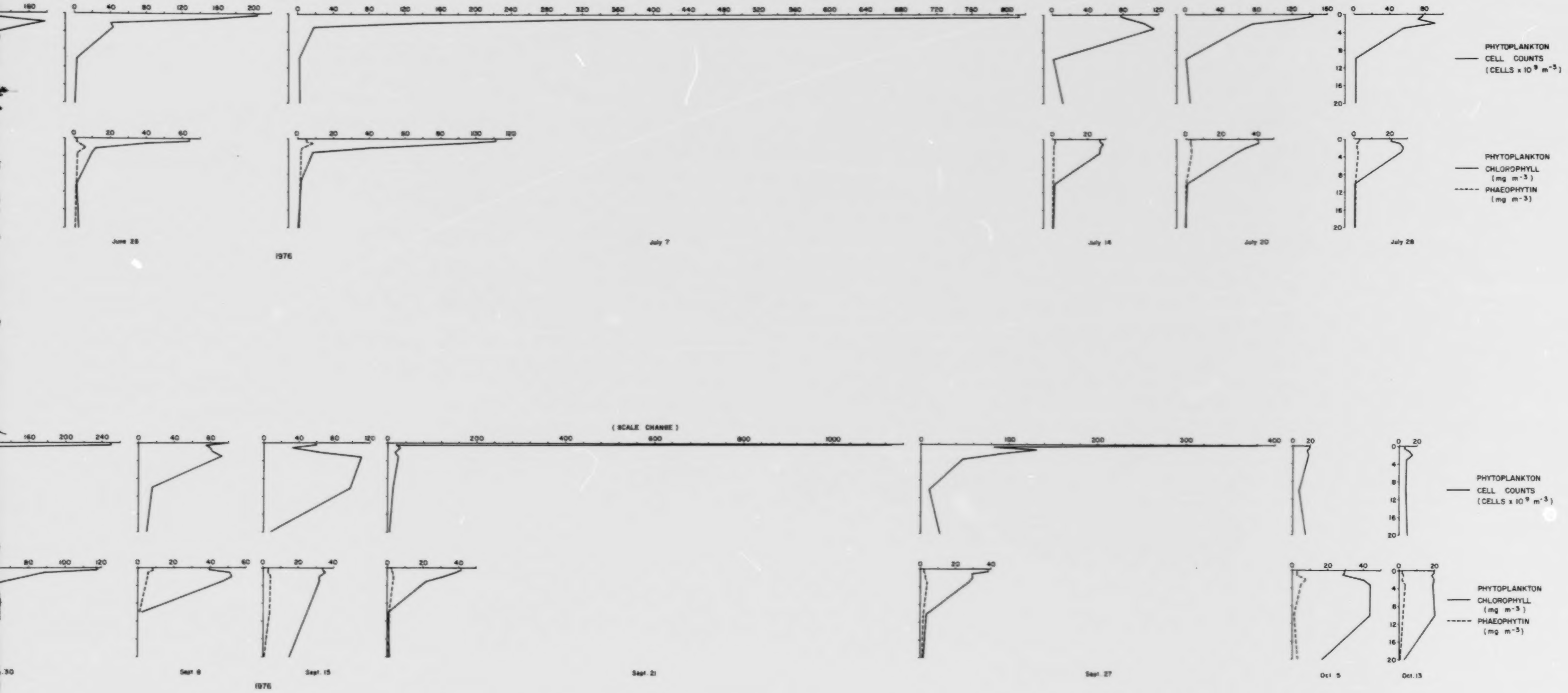

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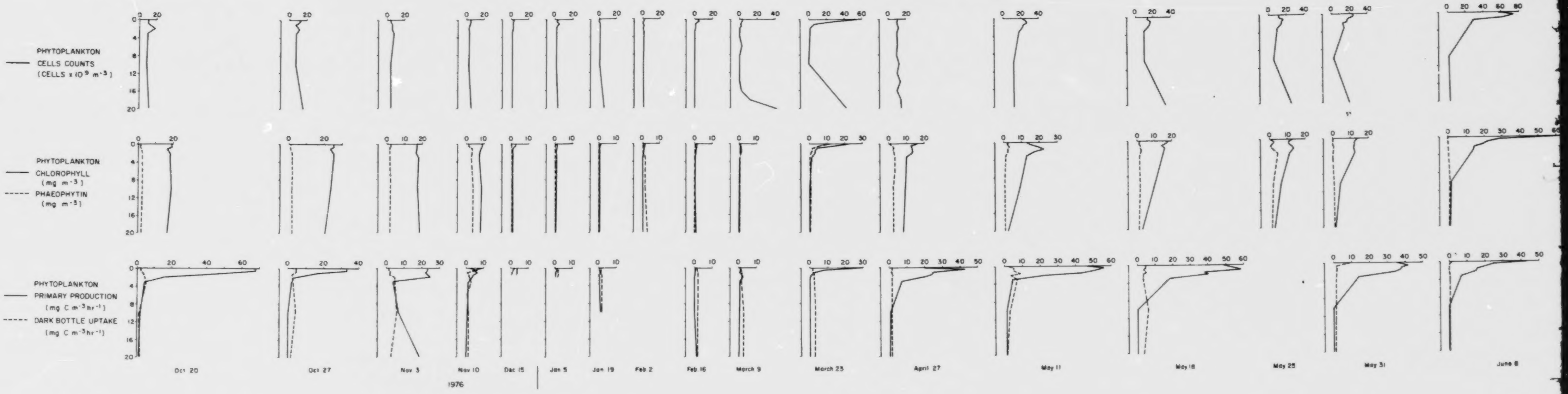
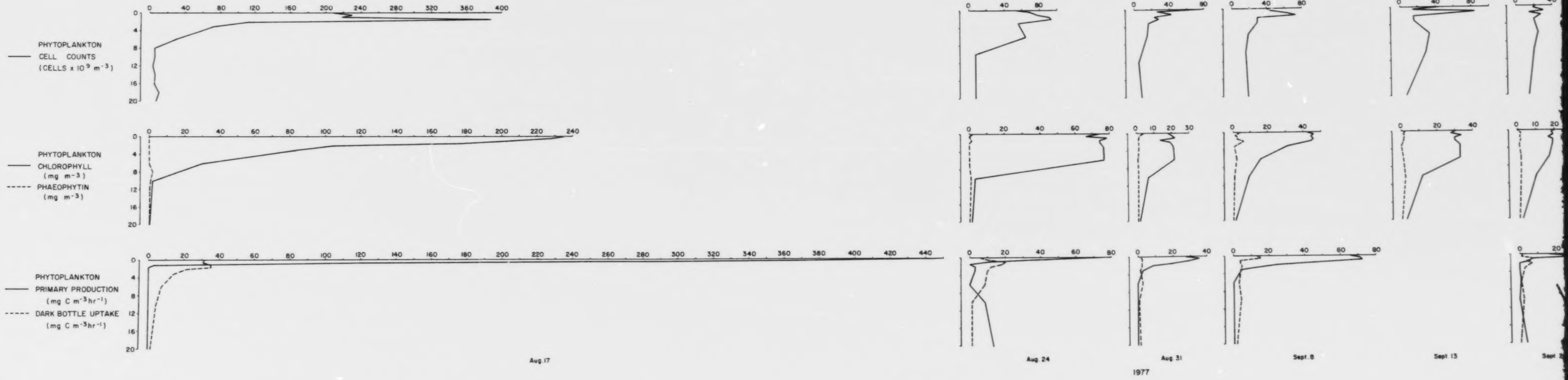
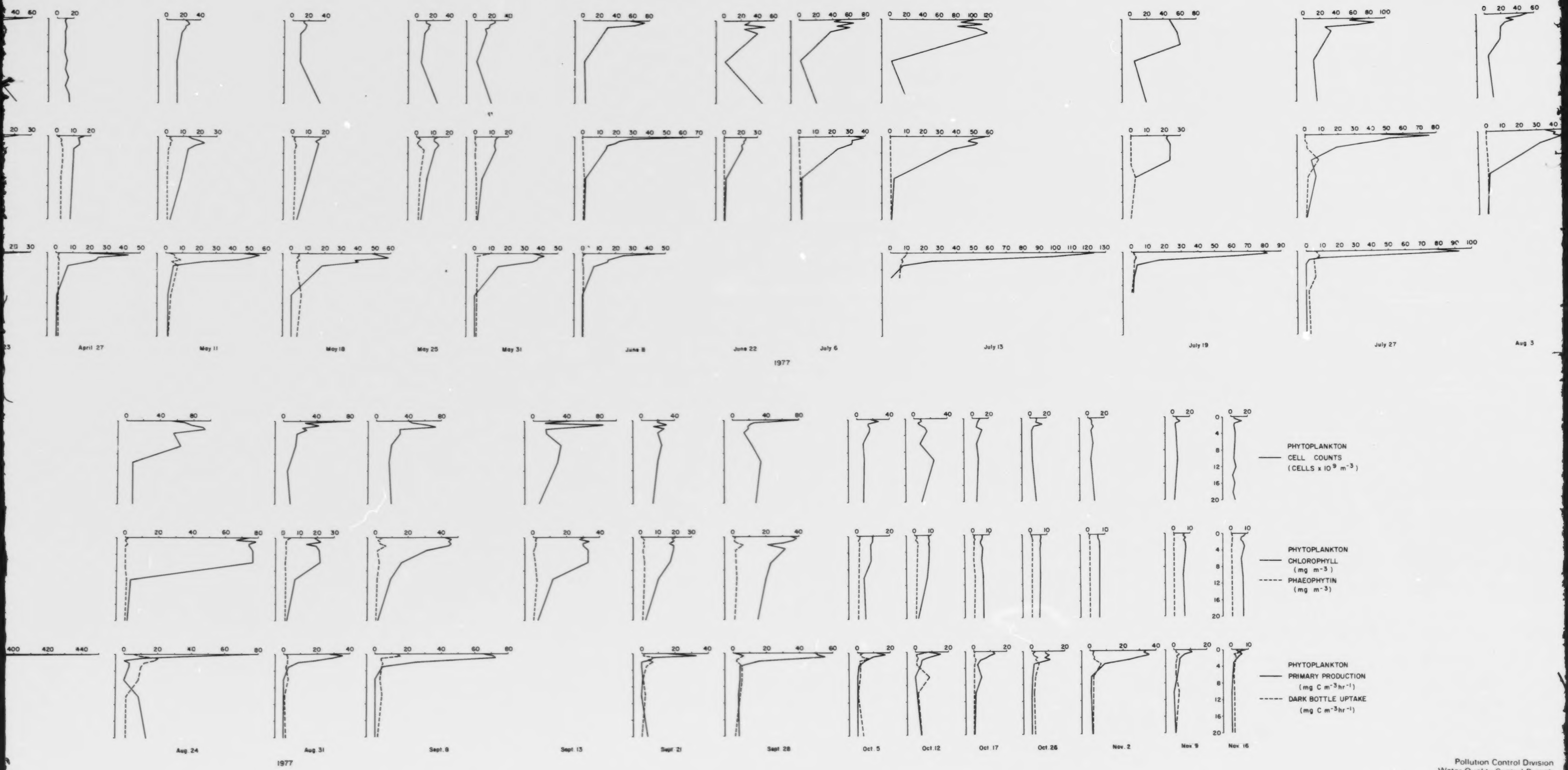
FIG.46 VERTICAL DISTRIBUTION OF PHYTOPLANKTON STANDING CROP IN BAPTISTE LAKE (SOUTH) FROM MAR. 16 TO OCT. 13, 1976


PHYTOPLANKTON
CELLS COUNTS
(CELLS x 10⁹ m⁻³)
PHYTOPLANKTON
CHLOROPHYLL
(mg m⁻³)
PHAEOPHYTIN
(mg m⁻³)
PHYTOPLANKTON
PRIMARY PRODUCTION
(mg C m⁻³hr⁻¹)
DARK BOTTLE UPTAKE
(mg C m⁻³hr⁻¹)
0
4
8
12
16
20
Oct 20
Oct 27
Nov 3
Nov 10
Dec 15
1976
Jan 5
Jan 19
Feb 2
Feb 16
March 9
March 23
April 27
May 11
May 18
May 25
May 31
June 8


PHYTOPLANKTON
CELL COUNTS
(CELLS x 10⁹ m⁻³)
PHYTOPLANKTON
CHLOROPHYLL
(mg m⁻³)
PHAEOPHYTIN
(mg m⁻³)
PHYTOPLANKTON
PRIMARY PRODUCTION
(mg C m⁻³hr⁻¹)
DARK BOTTLE UPTAKE
(mg C m⁻³hr⁻¹)
0
4
8
12
16
20
Aug.17
Aug.24
Aug.31
Sept.8
Sept.13
1977



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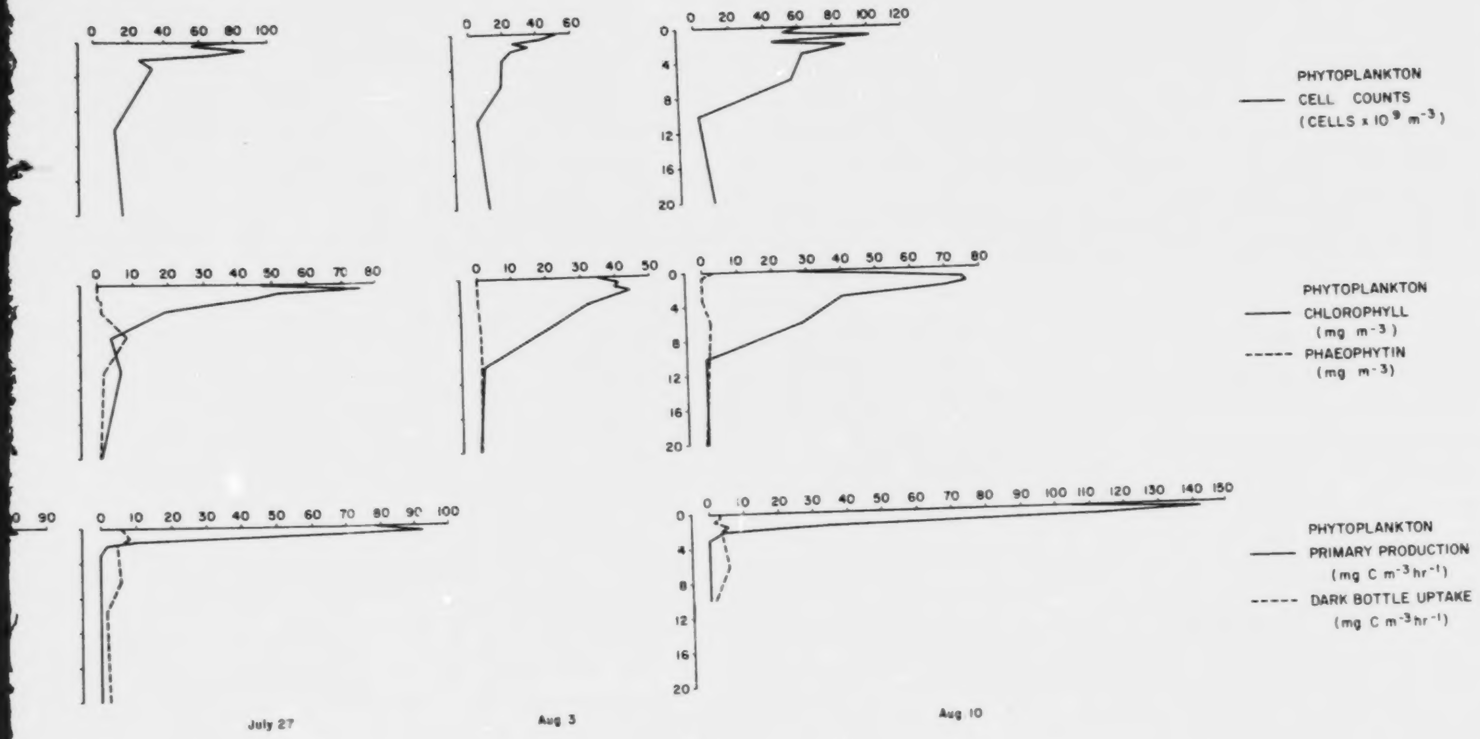
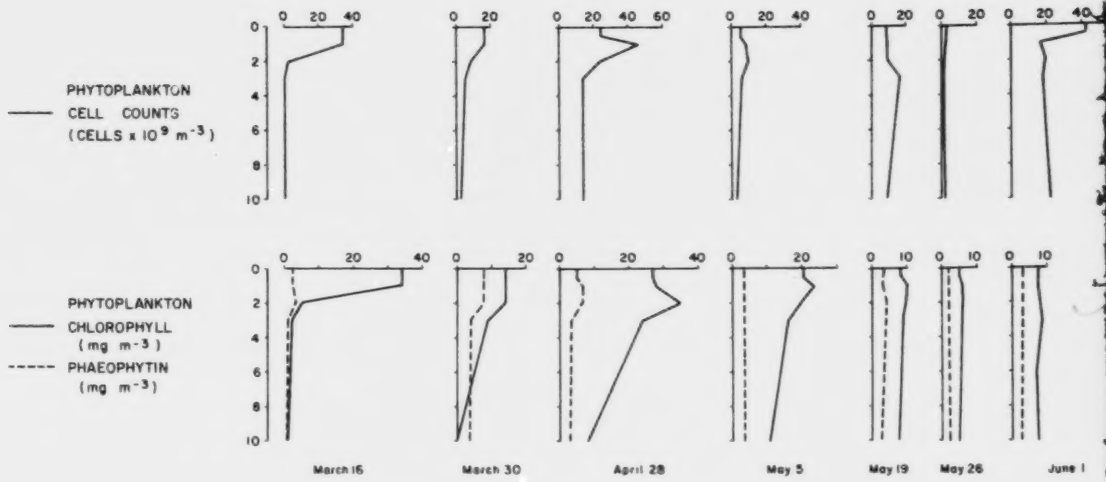
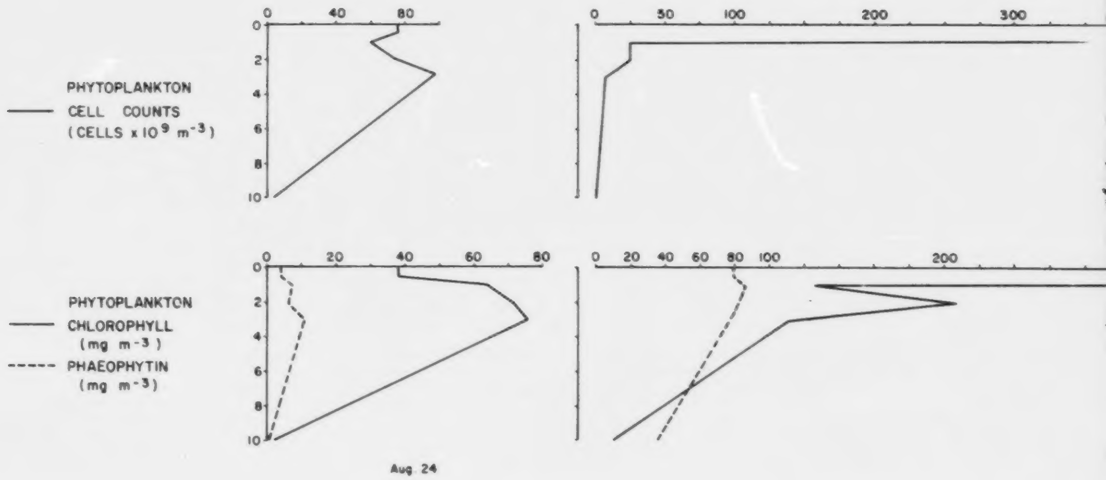
Water Quality Control Branch

FIG. 47 VERTICA
AND PRI
OCT. 20







PHYTOPLANKTON
CELL COUNTS
(CELLS x 10⁹ m⁻³)

PHYTOPLANKTON
CHLOROPHYLL
(mg m⁻³)
PHAEOPHYTIN
(mg m⁻³)

PHYTOPLANKTON
PRIMARY PRODUCTION
(mg C m⁻³hr⁻¹)
DARK BOTTLE UPTAKE
(mg C m⁻³hr⁻¹)

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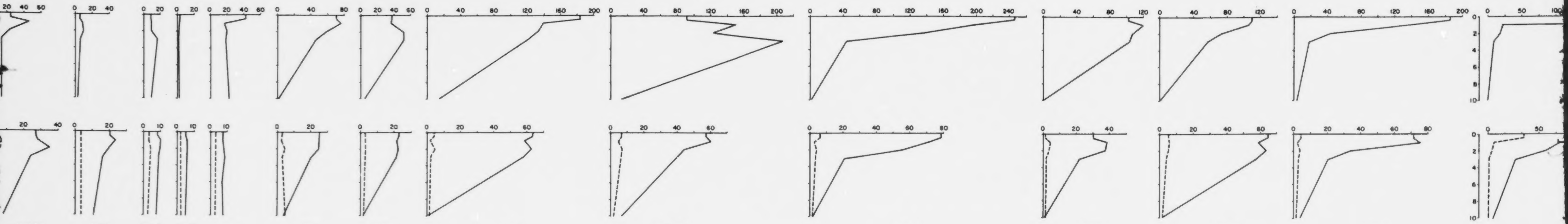
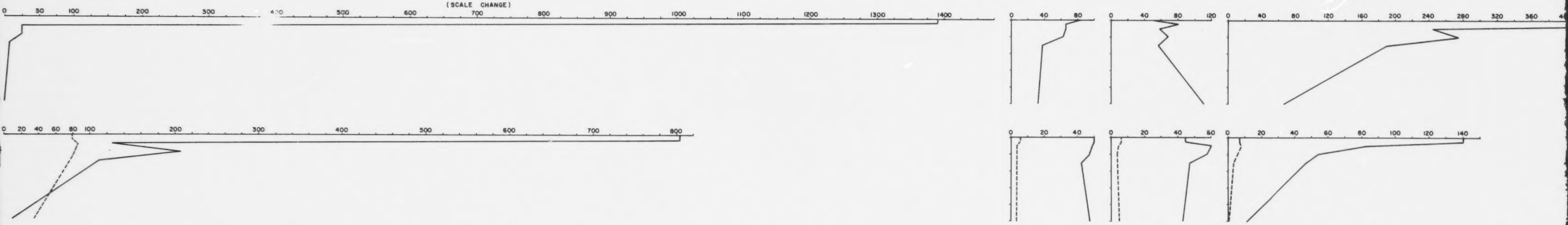
FIG. 47 VERTICAL DISTRIBUTION OF PHYTOPLANKTON STANDING CROP AND PRIMARY PRODUCTION IN BAPTISTE LAKE (SOUTH) FROM OCT. 20, 1976 TO NOV. 16, 1977



April 28 May 5 May 19 May 26 June 1 June 8 June 15 June 22 June 28 July 7 July 14 July 20 Aug. 2

1976



Aug. 30 Sept. 8 Sept. 15 Sept. 21

1976

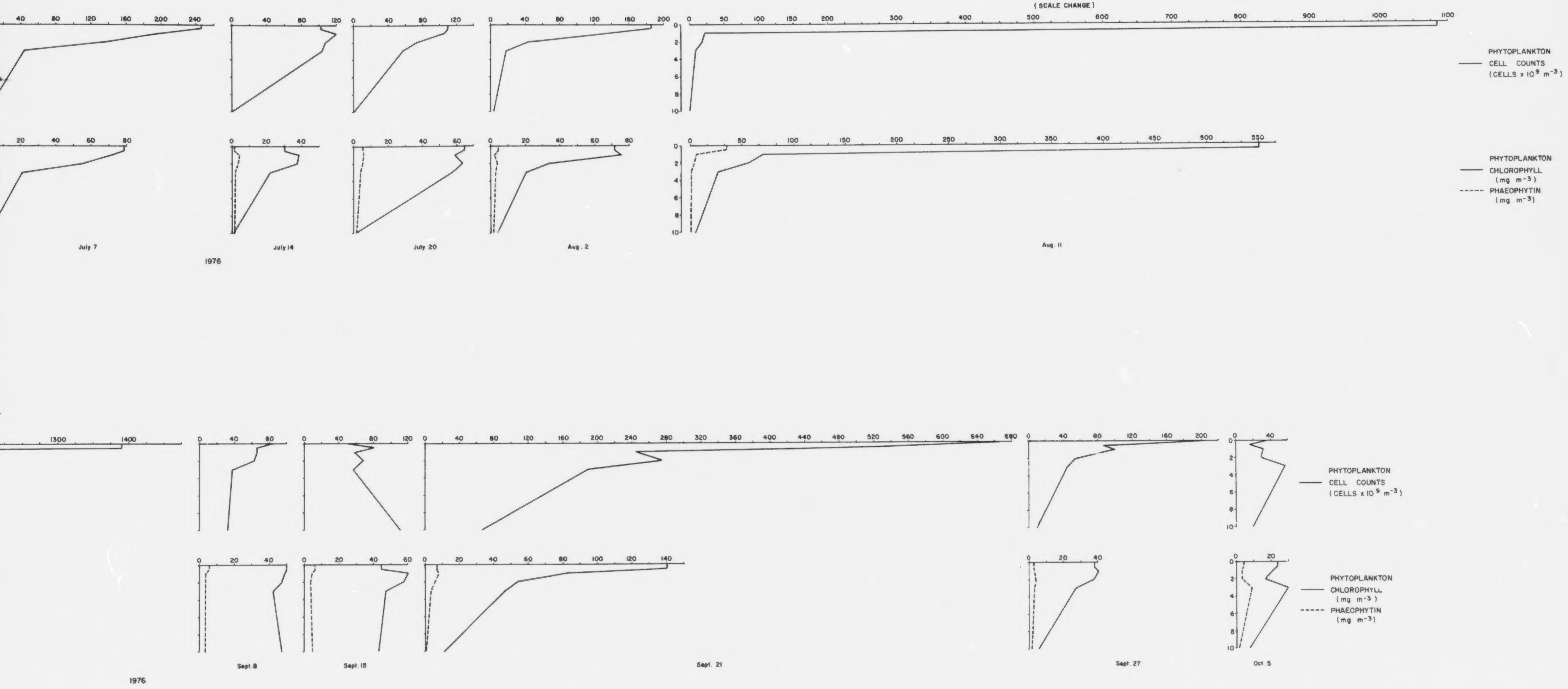


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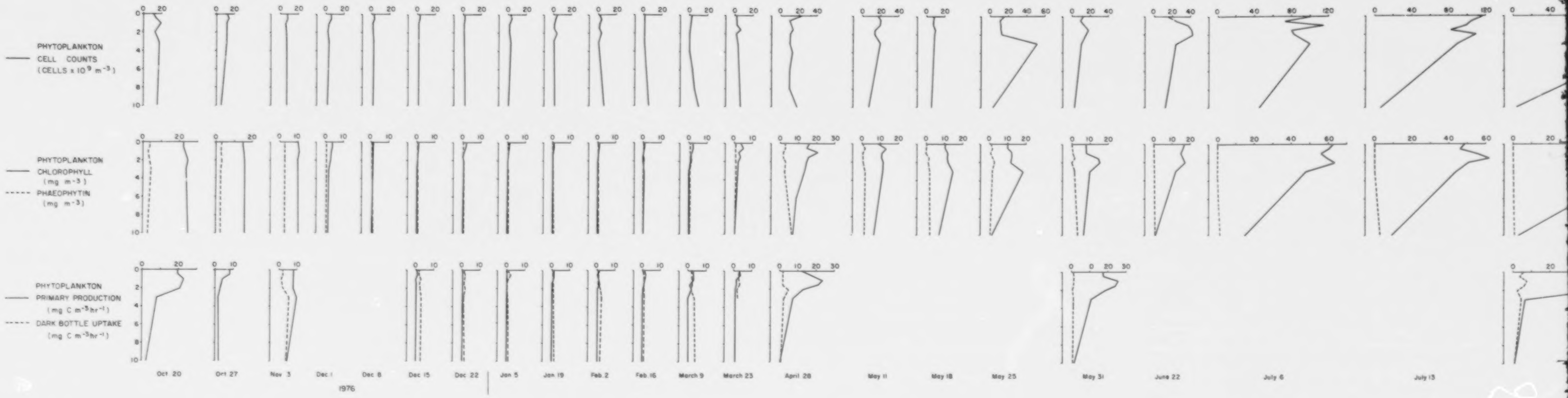
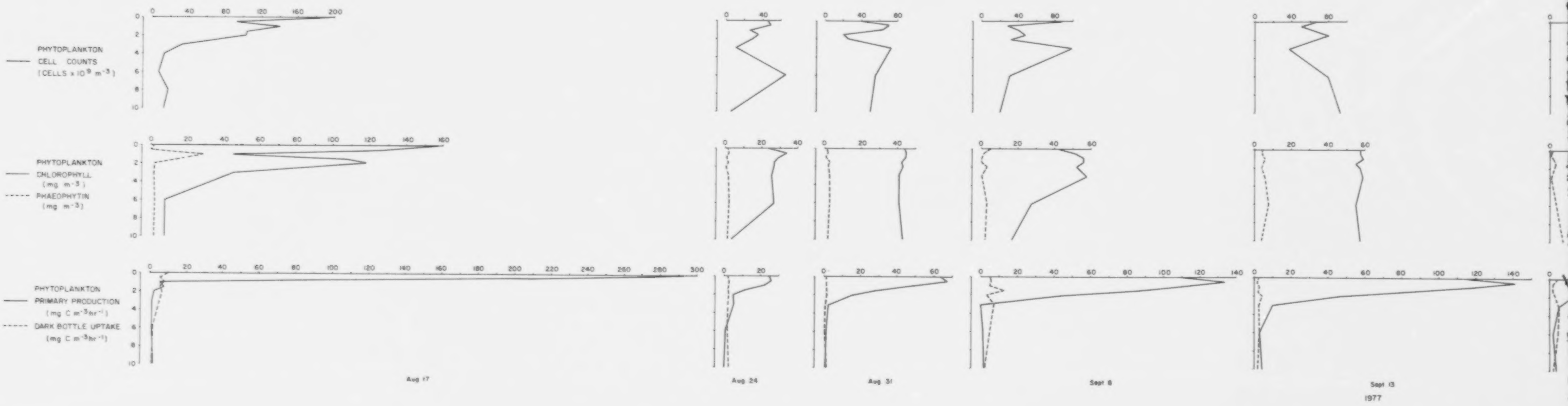
FIG. 48 VERTICAL DISTRIBUTION OF PHYTOPLANKTON STANDING CROP IN BAPTISTE LAKE (NORTH) FROM MAR. 16 TO OCT. 5, 1976


PHYTOPLANKTON CELL COUNTS (CELLS x 10^9 m^{-3})
PHYTOPLANKTON CHLOROPHYLL (mg m^{-3})
PHAEOPHYTIN (mg m^{-3})
PHYTOPLANKTON PRIMARY PRODUCTION (mg C m^{-3}hr^{-1})
DARK BOTTLE UPTAKE (mg C m^{-3}hr^{-1})
Oct 20
Oct 27
Nov 3
Dec 1
Dec 8
Dec 15
Dec 22
1976
Jan 5
Jan 19
Feb 2
Feb 16
March 9
March 23
April 28
May 11
May 18
May 25
May 31
June 22
July 6
July 13


PHYTOPLANKTON CELL COUNTS (CELLS x 10^9 m^{-3})
PHYTOPLANKTON CHLOROPHYLL (mg m^{-3})
PHAEOPHYTIN (mg m^{-3})
PHYTOPLANKTON PRIMARY PRODUCTION (mg C m^{-3}hr^{-1})
DARK BOTTLE UPTAKE (mg C m^{-3}hr^{-1})
Aug 17
Aug 24
Aug 31
Sept 8
Sept 13
1977

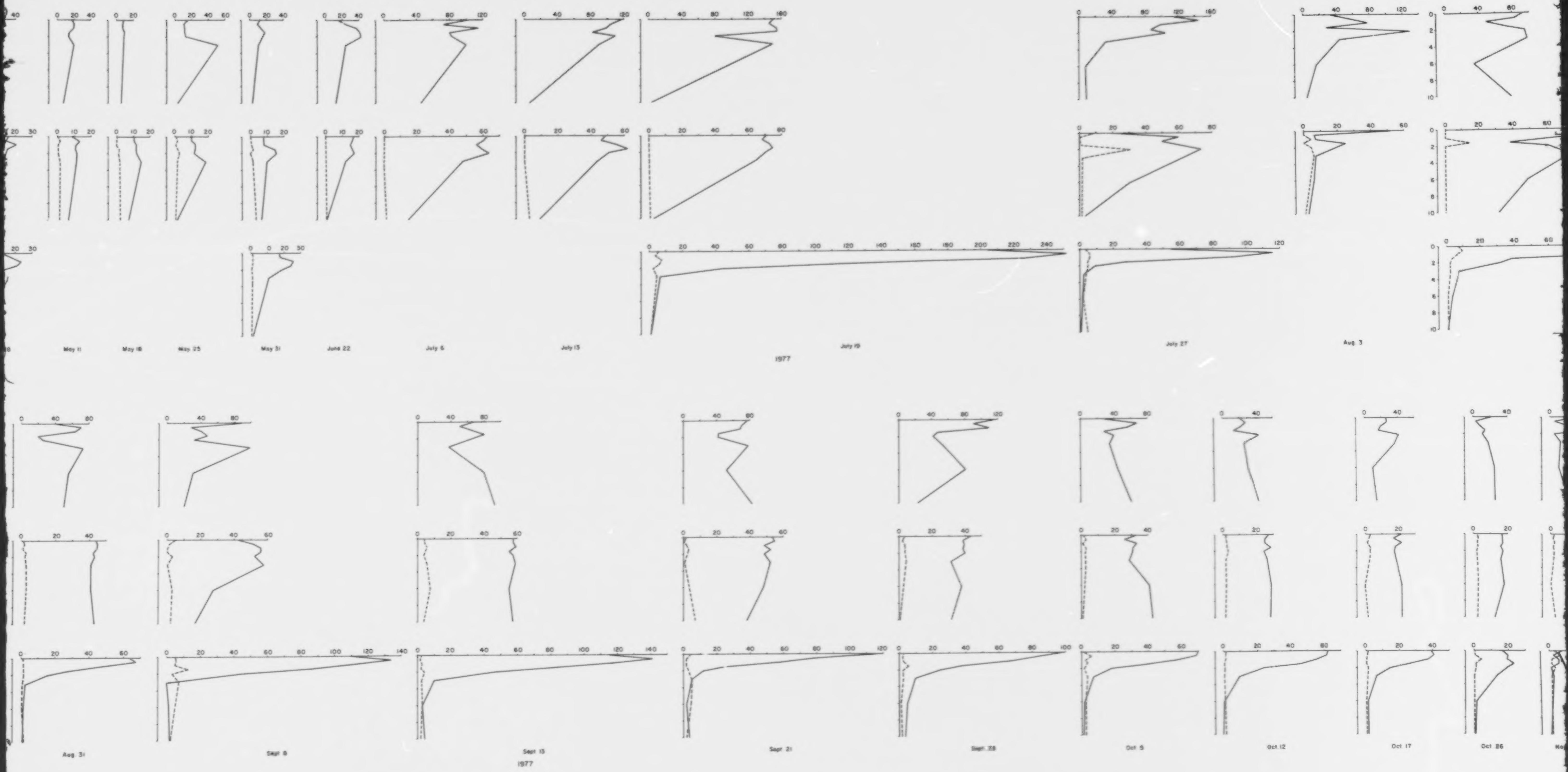


FIG. 49 VE
AN
OC

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Water Quality Control Branch

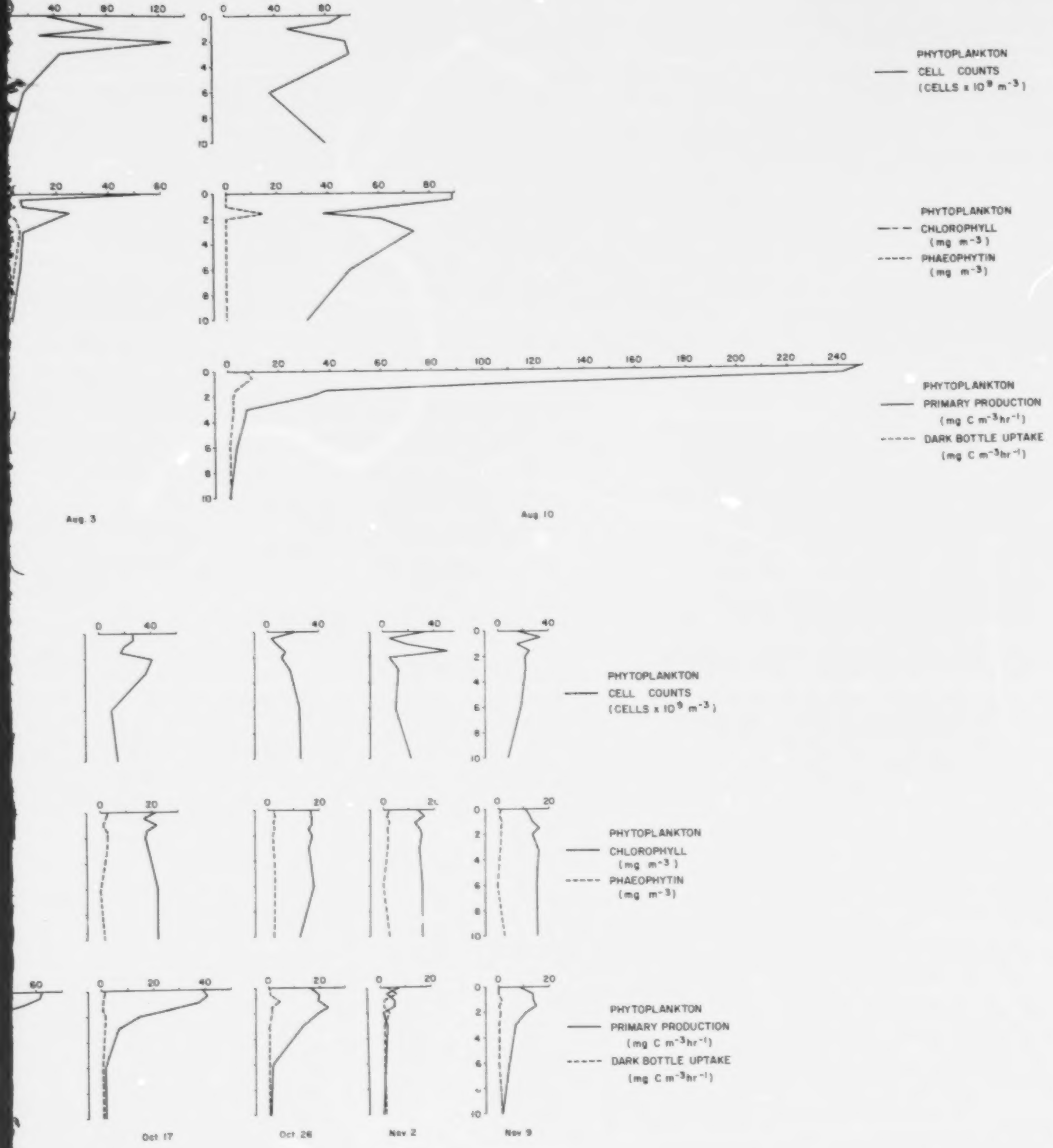

FIG. 49 VERTICAL DISTRIBUTION OF PHYTOPLANKTON STANDING CROP AND PRIMARY PRODUCTION IN BAPTISTE LAKE (NORTH) FROM OCT. 20, 1976 TO NOV. 9, 1977

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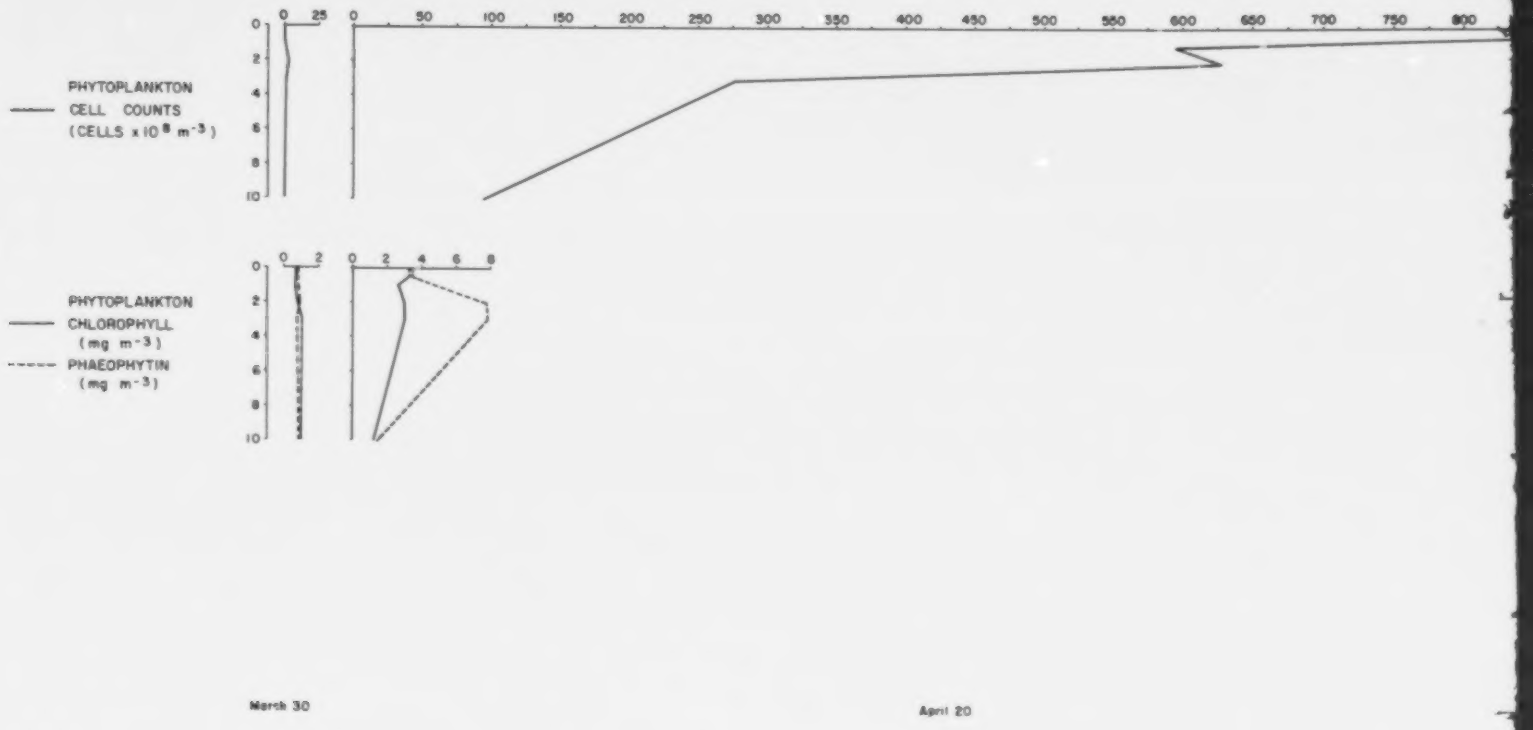
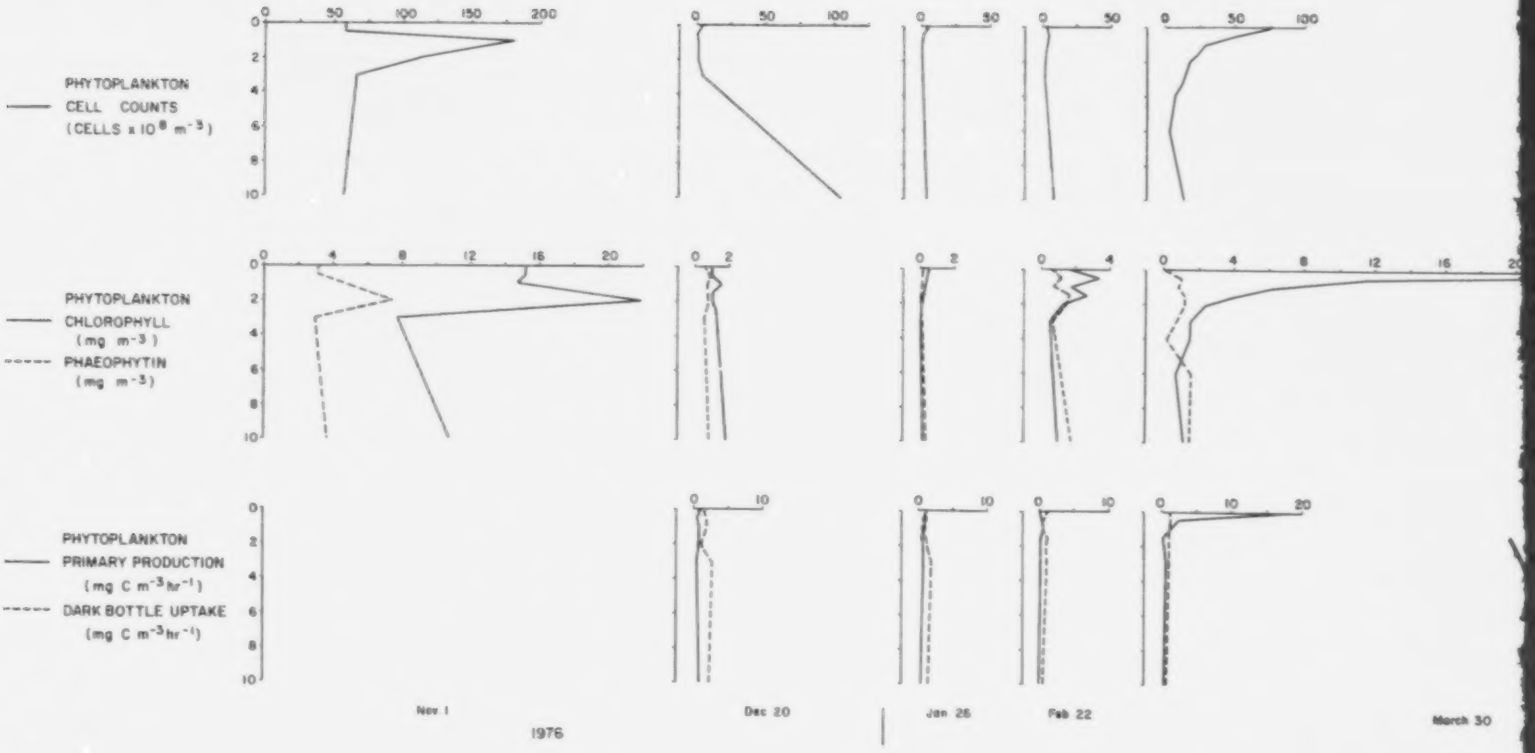

PHYTOPLANKTON
CELL COUNTS
(CELLS x 10⁸ m⁻³)
PHYTOPLANKTON
CHLOROPHYLL
(mg m⁻³)
PHAEOPHYTIN
(mg m⁻³)
0 25
0 50 100 150 200 250 300 350 400 450 500 550 600 650 700 750 800
0 2
0 2 4 6 8
0 2 4 6 8 10
March 30
April 20


PHYTOPLANKTON
CELL COUNTS
(CELLS x 10⁸ m⁻³)
PHYTOPLANKTON
CHLOROPHYLL
(mg m⁻³)
PHAEOPHYTIN
(mg m⁻³)
PHYTOPLANKTON
PRIMARY PRODUCTION
(mg C m⁻³hr⁻¹)
DARK BOTTLE UPTAKE
(mg C m⁻³hr⁻¹)
0 50 100 150 200
0 50 100
0 50
0 50
0 50 100
0 4 8 12 16 20
0 2
0 2
0 4
0 4 8 12 16 20
0 10
0 10
0 10
0 10 20
0 2 4 6 8 10
Nov 1
1976
Dec 20
Jan 26
Feb 22
March 30

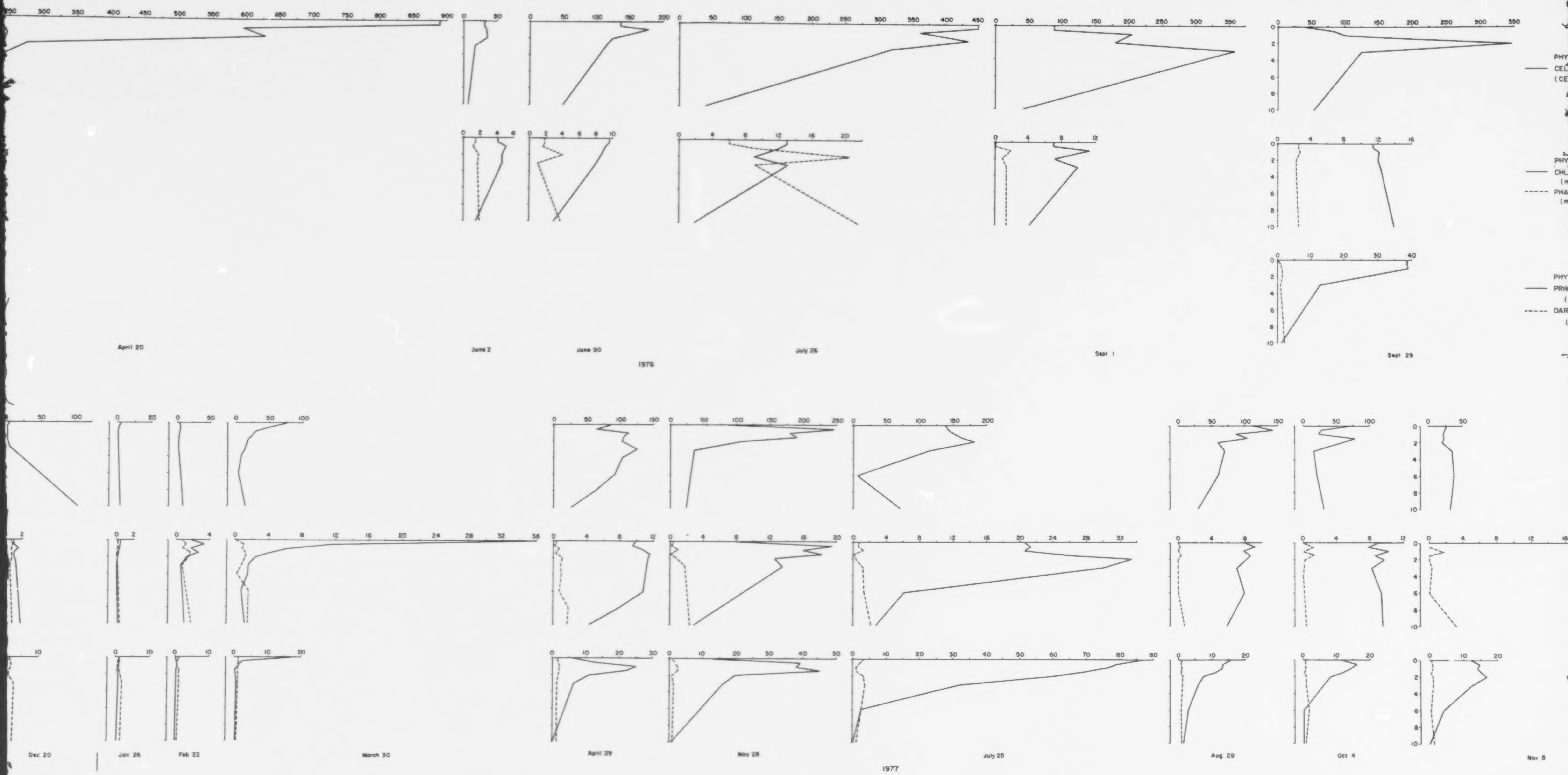


FIG.50 VERTICAL DISTRIBUTION OF PHY
AND PRIMARY PRODUCTION IN
TO NOV. 8, 1977

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Water Quality Control Branch

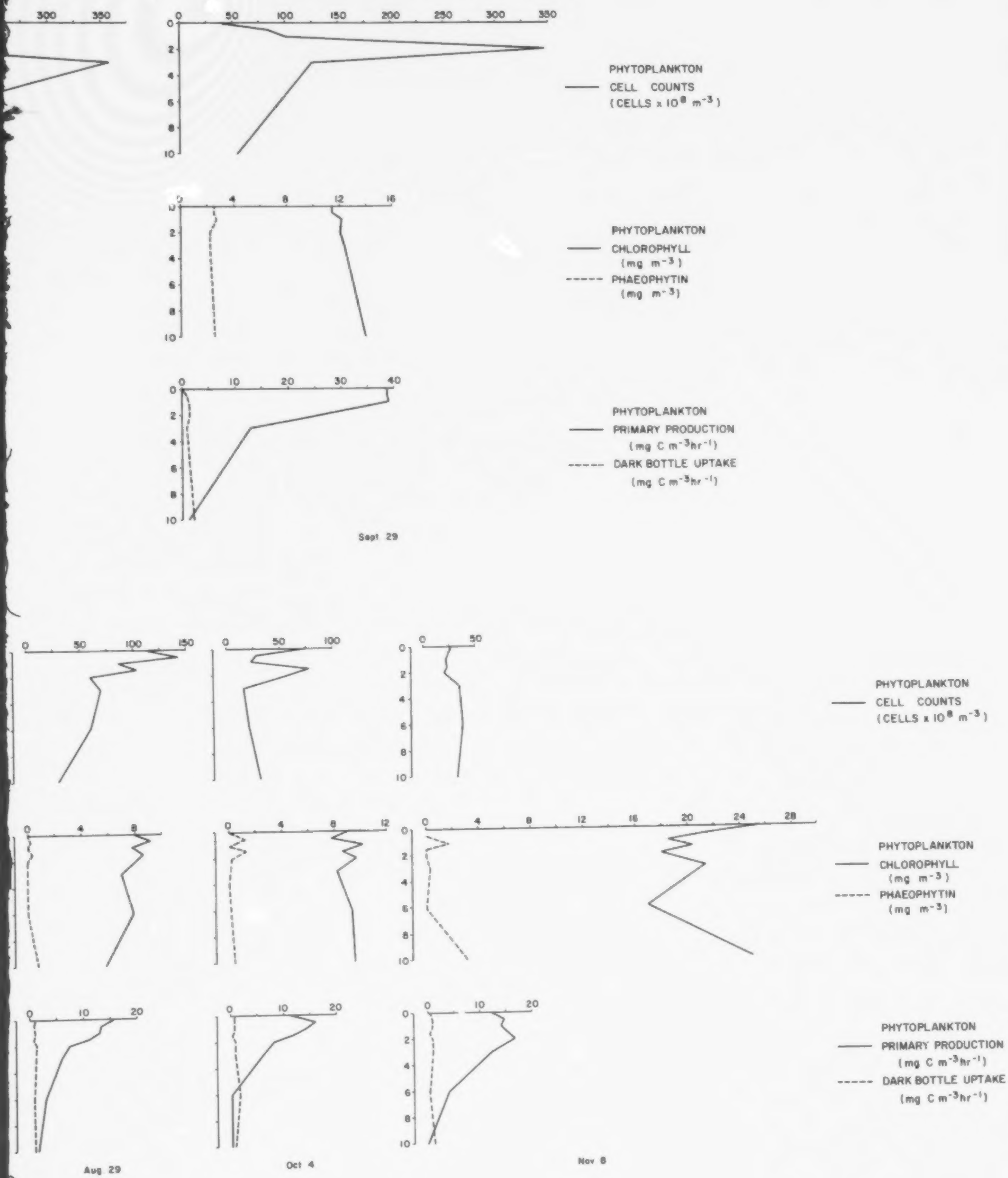
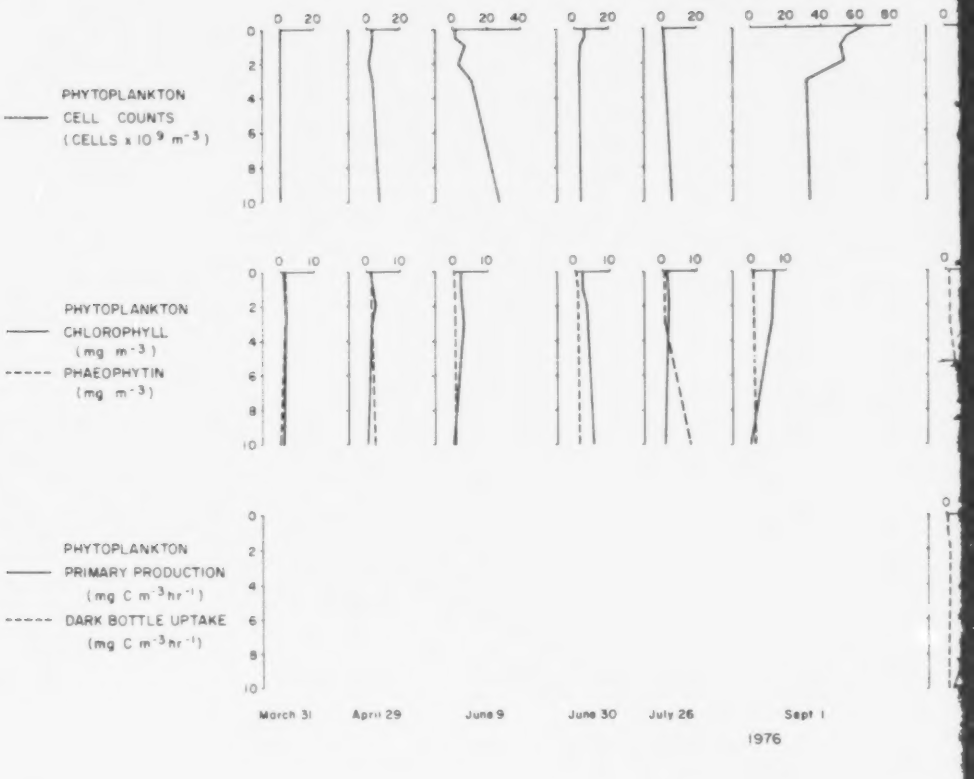


FIG.50 VERTICAL DISTRIBUTION OF PHYTOPLANKTON STANDING CROP AND PRIMARY PRODUCTION IN GHOST LAKE FROM MAR.30, 1976 TO NOV.8, 1977



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Water Qual

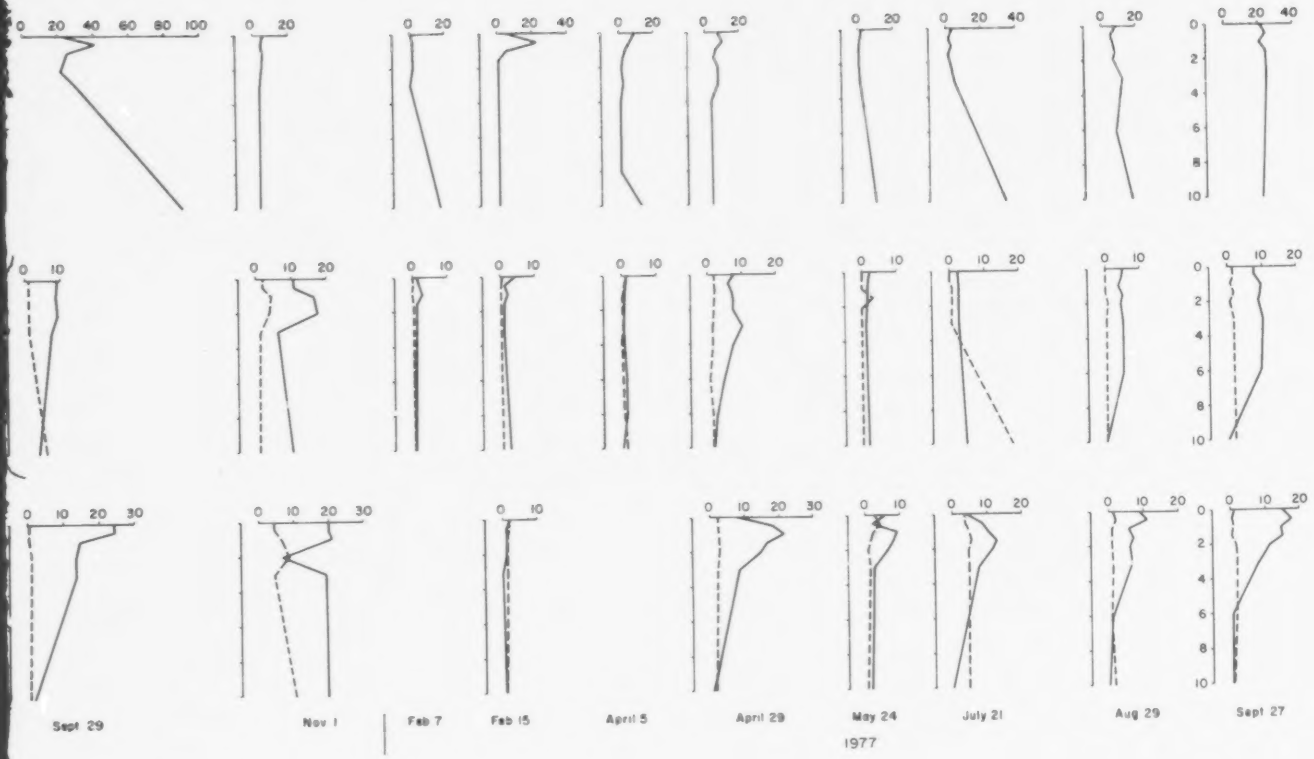

FIG.51 VERTICAL DISTRIBUTION OF PHYTOPLANKTON STANDING CROP AND PRIMARY PRODUCTION IN ISLAND LAKE FROM MAR. 31, 1976 TO SEPT. 27, 1977

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Quality Control Branch

of highest water temperatures in many lakes (Hutchinson 1967), during which it encysts in the bottom sediments. Chymko (1977) suggested that this was the reason for this species' disappearance in Lesser Slave Lake, and is likely the reason for its absence in August in Baptiste. Samples from Island Lake were taken too infrequently to determine whether a similar disappearance occurred.

Other important species of cyclopoid copepod in Baptiste during 1976 - 1977 were Acanthocyclops vernalis and Mesocyclops edax. A. vernalis, similar in size to Diacylops, is likely to compete with it for prey. Although observed in all three lakes, it was most abundant in Ghost. This species is separated temporally to some extent from Diacyclops; in Baptiste it was first observed in June in both basins in 1976. Its maximum population occurred during the warm water period.

M. edax was never abundant in any of the lakes. Its larger size likely reduces competition with the other two dominant species. All three prey on cladocerans and juvenile copepods, including members of their own species.

The dominant calanoid copepod in all three lakes was Diaptomus oregonensis. Also observed on two occasions in Ghost Lake was Acanthodiaptomus denticornis. D. oregonensis was present in relatively high numbers on all sampling dates in Island, and on most sampling dates in Ghost, but was absent from late winter samples in Baptiste. This species is very common in Alberta lakes, and is often the only calanoid species in the lake in which it occurs.

7.10.2 CLADOCERA

Several species of Cladocera were abundant in the three lakes during 1976 - 1977. In Baptiste, Daphnia galeata mendotae was common throughout the summer, while Chydorus sphaericus became numerous after July. Populations of Daphnia peaked in July and August in the north basin, whereas in the south basin this peak occurred much later. Males were observed in September and October in both basins; the species reproduces parthenogenetically until unfavorable conditions arise. Such conditions might be a reduction in food supply, at which time males are assexually produced, mate with the females, and overwintering eggs are produced.

Chydorus inhabits littoral areas until midsummer and move into the plankton when blue-green algae become abundant. According to workers cited in Hutchinson (1967) the changing light conditions brought about by algal blooms induce this habitat change in Chydorus.

These two species were present in Ghost and Island Lakes; Daphnia galeata mendotae was very abundant in late fall 1976 in both lakes while Chydorus occurred in small numbers. The smaller population of Chydorus in the plankton of these lakes may be related to the lower productivities of the latter.

A less dominant species of Daphnia in the three lakes was D. pulicaria. Its maximum population in Island Lake occurred in May-June, prior to the peak of its probable competitor, D. galeata mendotae. Males were observed in the population at this time, after which the species was not observed in samples. In contrast it was present in late summer in Ghost Lake although it resembled the slightly helmeted species D. schoedleiri. The typical D. pulicaria form was present before and after this period. D. pulicaria was less abundant in the north basin of Baptiste.

Other cladocerans of less numerical importance in these lakes included: Diaphanosoma leuchtenbergianum, which occurred during maximum temperatures; Ceriodaphnia lacustris, which was especially abundant in Ghost Lake in June and in Island Lake in late August; and Bosmina longirostris, which was most abundant from late fall through spring when water temperatures were low.

7.10.3 ROTATORIA

The greatest number of species of rotifers were observed in Island Lake (22 species) and the fewest in Baptiste Lake (15 species). However, population numbers appeared to be highest in Baptiste especially during spring and late fall. A large biomass with few species tends to indicate more eutrophic conditions (Gannon and Stemberger 1978).

Dominants in Baptiste included the perennial species Keratella quadrata, K. cochlearis (including K. earlinae), and several species of Polyarthra. Kellicottia longispina and Conochilus sp. were also very abundant at times.

All of these species were present in Ghost and Island. K. cochlearis was most abundant in mid-summer during both sampling years in Ghost and Island Lakes; in contrast, this species complex was more abundant in spring and fall in Baptiste.

Baker (1979) who worked on Hastings Lake near Edmonton, found that three species of Keratella, including K. cochlearis and K. quadrata also had maximal populations in spring and autumn. Birth rates of these species were positively correlated with certain small algae, but were not correlated with species of blue-green algae, which were abundant during midsummer as in Baptiste. Other rotifer species were positively correlated with bacteria, also a good food source for these animals.

7.11 BACTERIAL DISTRIBUTION

Five surface surveys on Baptiste Lake and two surface surveys on Ghost and Island Lakes were conducted to determine the presence and distribution pattern for total and faecal coliform bacteria. Samples from Baptiste Lake were taken from a grid of 53 surface points on the following dates: May 31, June 16, July 5, August 15, and September 7, 1976. Ghost and Island Lakes were sampled less intensively (2 and 10 samples respectively) on the following dates: July 26 and September 1, 1976.

In Baptiste Lake, detectable levels of both total coliform and faecal coliform bacteria were found on all survey dates (Figures 52 and 53). The distribution of bacteria did not, however, display any consistent pattern from one sample date to the next, and showed no relationship to the location of cottages. In general, the sites of the highest bacterial concentrations proved to be either off-shore or adjacent to essentially undeveloped shoreline areas: waterfowl frequenting these areas may have been the principal source.

The upper limit for the 5 tube dilution technique as used by the Provincial Laboratory of Public Health, was 16 total coliforms or faecal coliforms per 100 mL. In order to determine absolute numbers of coliform bacteria, which frequently exceeded these limits, an extra set of six samples were taken on June 23, 1976, and analyzed by the membrane filter technique (Figure 54). In three samples the levels exceeded the upper limits of the five tube dilution technique, and the range for all samples was 6 - 68 coliforms per 100 mL, which is within the range reported to occur in certain lakes in the National Parks of Western Canada (Anderson and Krochak 1972). Based on these miscellaneous samples, Baptiste Lake may be classed as suitable for direct contact recreation according to the Alberta Surface Water Quality Objectives.

Only two sampling sites were chosen for Ghost Lake, the watershed of which is essentially unpopulated. One of these sites was adjacent to a campground on the north-western end of the lake and one adjacent to our routine sampling site (Figure 55). No bacteria of either type were detected from either site, for both dates, for this lake. This may reflect the undeveloped nature of this lake but it may also be influenced by the more exposed, less vegetated nature of the sites chosen (less suitable for waterfowl).

A total of ten surface sample points were chosen for Island Lake. The lower 8 sites were adjacent to shoreline development along the western shores (Figure 56). Of the two other sites, one was adjacent to an abandoned farm site at the entrance to the northern bay and one was at our routine sample site in that bay. Generally, higher bacterial concentrations were found adjacent to shoreline development and this may reflect the influence of waterfowl frequenting the extensive littoral zone of that region. Our routine sampling site within the bay displayed low or undetectable levels, resembling those of Ghost and mid-lake Baptiste stations.

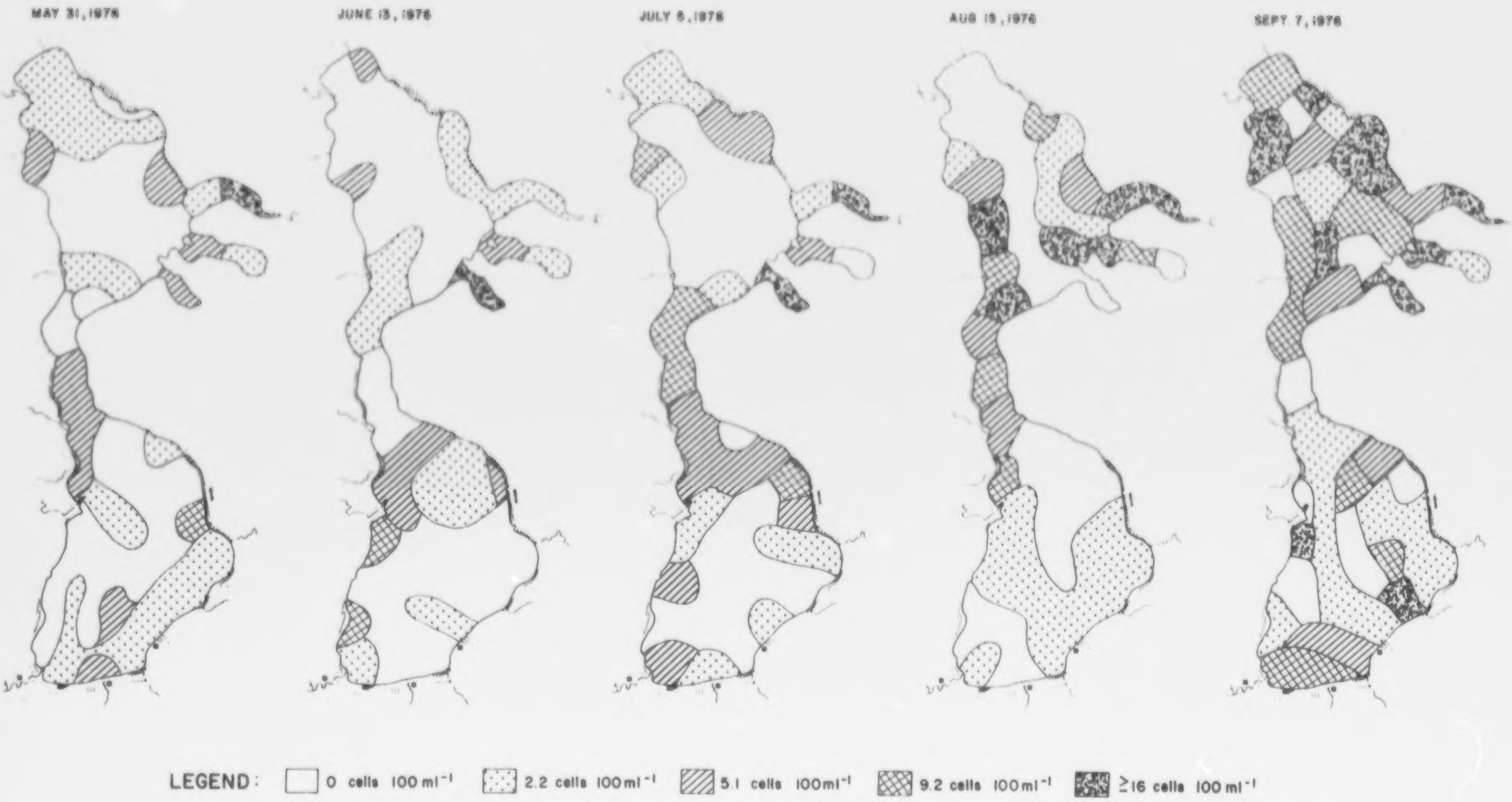
MAY 31, 1976
JUNE 13, 1976
JULY 6, 1976
AUG 18, 1976
SEPT. 7, 1976
LEGEND:
0 cells 100 ml⁻¹
2.2 cells 100 ml⁻¹
5.1 cells 100 ml⁻¹
9.2 cells 100 ml⁻¹
≥16 cells 100 ml⁻¹

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FIG. 52 SURFACE DISTRIBUTION (0 m) OF TOTAL COLIFORM BACTERIA IN BAPTISTE LAKE

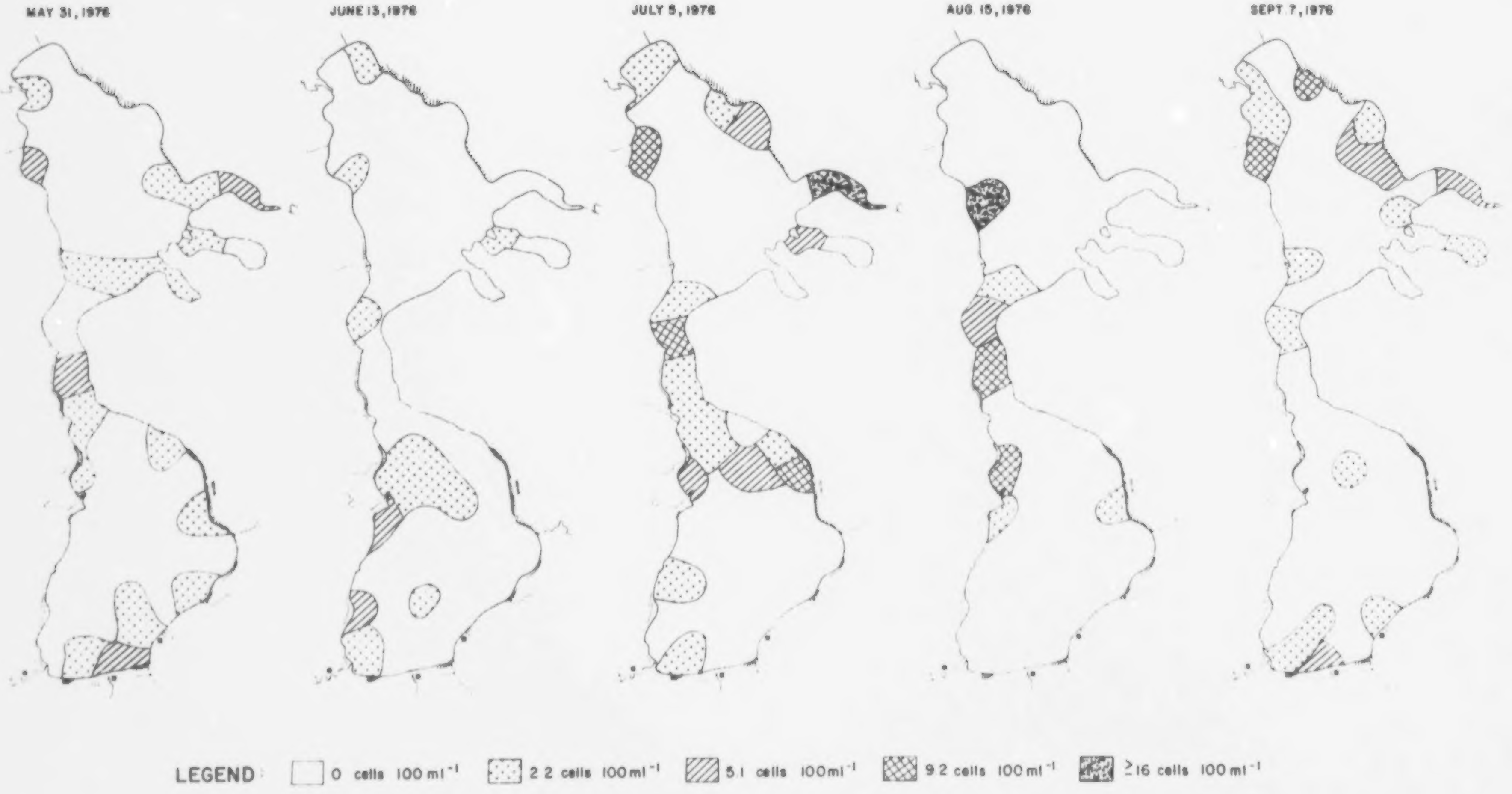
MAY 31, 1976
JUNE 13, 1976
JULY 5, 1976
AUG. 15, 1976
SEPT. 7, 1976
LEGEND:
0 cells 100 ml⁻¹
2.2 cells 100 ml⁻¹
5.1 cells 100 ml⁻¹
9.2 cells 100 ml⁻¹
≥16 cells 100 ml⁻¹

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FIG. 53 SURFACE DISTRIBUTION (0m) OF FAECAL COLIFORM BACTERIA IN BAPTISTE LAKE

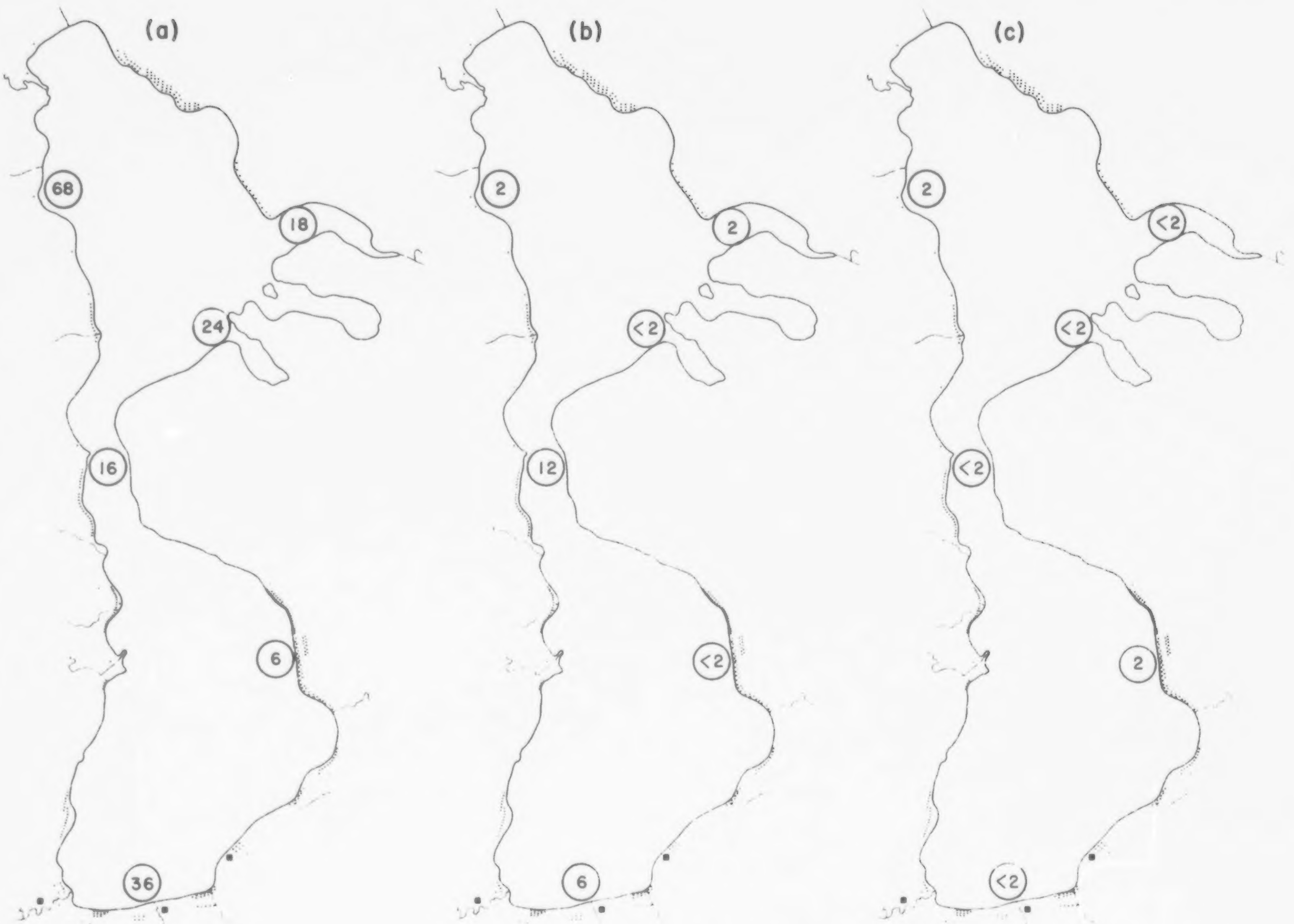
(a)
68
18
24
16
6
36
(b)
2
2
<2
12
<2
6
(c)
2
<2
<2
<2
2
<2

FIG.54 SURFACE DISTRIBUTION (0 m) OF (a) TOTAL COLIFORM, (b) FAECAL COLIFORM, AND (c) FAECAL STREPTOCOCCI BACTERIA IN BAPTISTE LAKE ON JUNE 23, 1976

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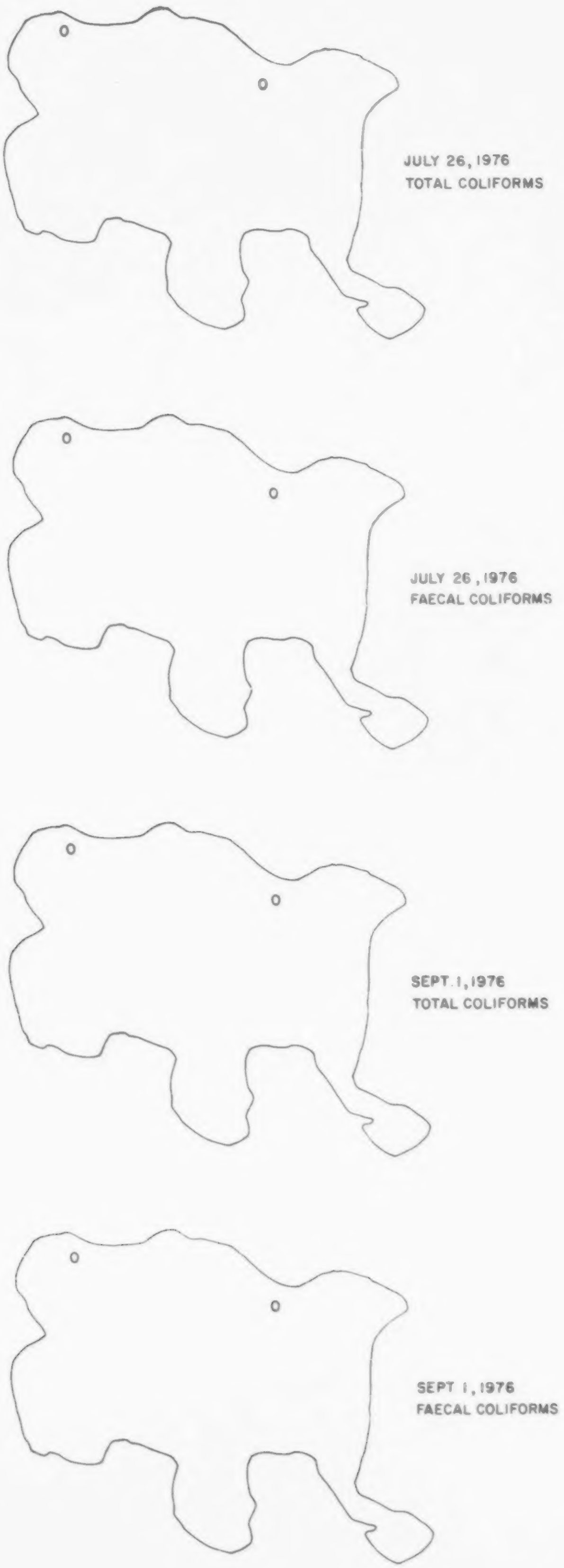
0
0
JULY 26, 1976
TOTAL COLIFORMS
0
0
JULY 26 , 1976
FAECAL COLIFORMS
0
0
SEPT. 1, 1976
TOTAL COLIFORMS
0
0
SEPT 1, 1976
FAECAL COLIFORMS

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FIG.55 SURFACE DISTRIBUTION (0 m) OF TOTAL AND FAECAL COLIFORM BACTERIA IN GHOST LAKE

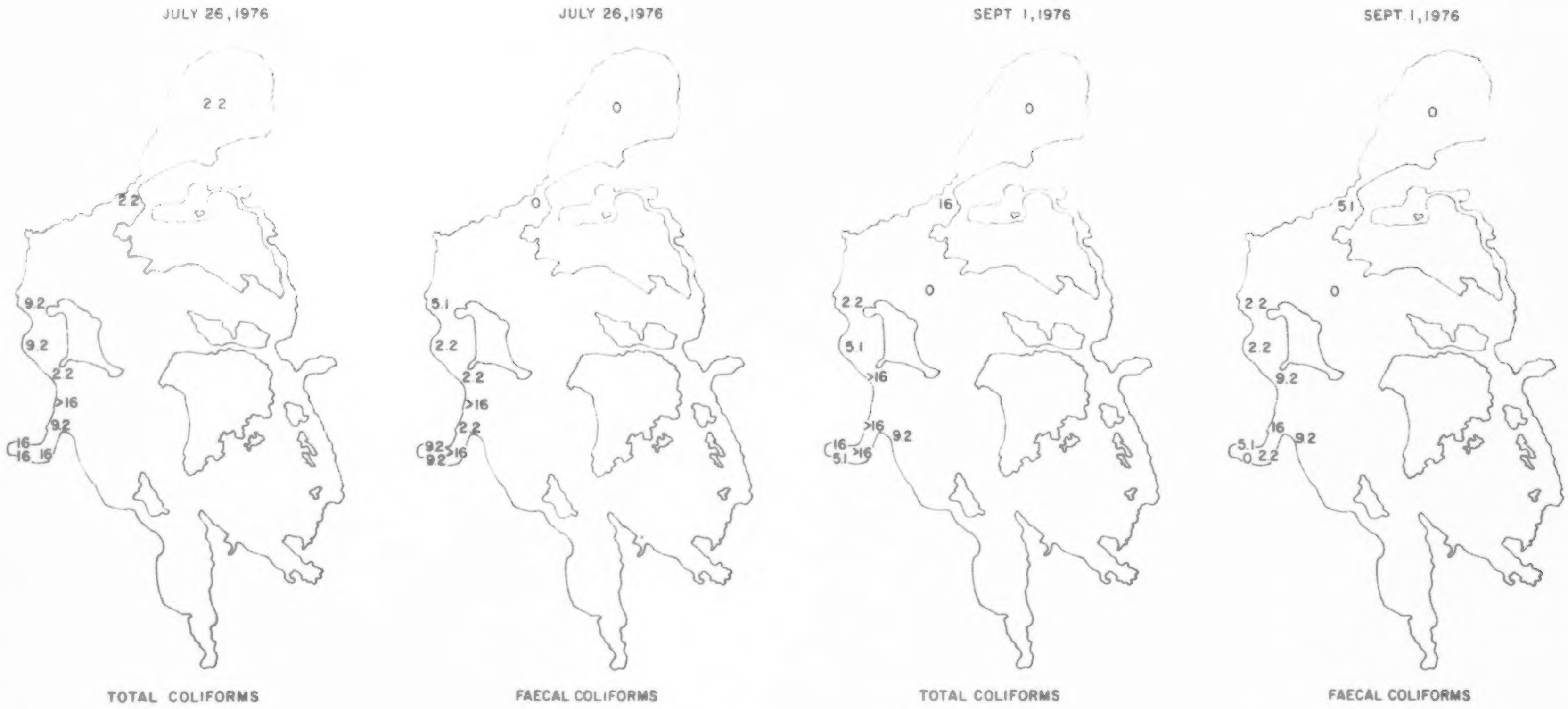
JULY 26, 1976
2.2
2.2
9.2
9.2
2.2
>16
9.2
16
16
16
TOTAL COLIFORMS
JULY 26, 1976
0
0
5.1
2.2
2.2
>16
2.2
9.2
9.2
>16
FAECAL COLIFORMS
SEPT 1, 1976
0
16
0
2.2
5.1
>16
>16
9.2
16
5.1
>16
TOTAL COLIFORMS
SEPT. 1, 1976
0
5.1
0
2.2
2.2
9.2
16
9.2
5.1
0
2.2
FAECAL COLIFORMS

FIG. 56 SURFACE DISTRIBUTION (0 m) OF TOTAL AND FAECAL COLIFORM BACTERIA IN ISLAND LAKE

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8.0 NITROGEN AND PHOSPHORUS LOADINGS

The potential inputs of phosphorus and nitrogen to a lake system are from the following sources:

a) surface runoff
b) groundwater seepage
c) atmospheric deposition (dustfall and precipitation)
d) sediment release, mixing processes
e) lakeshore development, point sources
f) other sources (leaf litter, waterfowl, N-fixation)

8.1 SURFACE RUNOFF

A three year summary of data describing the transport of total phosphorus and total nitrogen through surface runoff in the Baptiste Lake watershed is given in Tables 34 and 35, 36, and 37. Data for the transport of soluble reactive phosphorus, ammonia, nitrate-nitrite and organic nitrogen loadings are presented in the Appendix in Tables A25- A32. For diffuse runoff areas the export coefficients extrapolated from stream basins are shown in brackets.

Numerous studies are reported in the literature in which the export (mass per unit time) of nutrients is calculated from stream discharge and measured concentrations (Schindler et al. 1976; Bargh 1977, 1978; Johnson et al. 1969; Sharpley 1976; Ongley et al. 1977). The mass of nutrients exported from a given type of land use generally depends upon the hydrological regime, consequently long-term hydrological data are useful in characterizing the export patterns through watersheds (Minns and Johnson 1979; Schindler et al. 1976).

Table 34. Total phosphorus loading (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1}) calculated for stream basins.

Stream	1976	1977	1978	Mean Coefficient
A	21.21 (0.03)	307.52 (0.36)	223.22 (0.26)	0.22
B	5.13 (0.08)	129.51 (2.10)	31.52 (0.51)	0.90
C	45.57 (0.17)	109.69 (0.41)	68.22 (0.25)	0.28
D	31.51 (0.05)	56.62 (0.09)	109.06 (0.16)	0.10
E	888.73 (0.12)	500.99 (0.07)	1810.07 (0.25)	0.15
F	376.74 (0.12)	487.52 (0.15)	824.74 (0.25)	0.17
G	3.10 -	3.39 -	4.01 -	-
H	22.92 (0.34)	4.56 (0.07)	3.99 (0.06)	0.16
I	57.55 (0.16)	85.25 (0.24)	41.44 (0.16)	0.19
J	9.49 (0.09)	115.10 (1.03)	13.13 (0.12)	0.41
K	36.23 (0.15)	36.84 (0.15)	6.10 (0.02)	0.11
L	204.45 (0.03)	476.64 (0.07)	1469.29 (0.22)	0.11
M	5.96 (0.10)	22.49 (0.36)	21.56 (0.35)	0.27
N	164.14 (0.38)	105.64 (0.25)	158.71 (0.37)	0.33
O	391.52 -	1036.88 -	2122.39 -	-
Total A-M	1708.59	2336.12	46265.34	

Table 35. Total phosphorus loadings (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1}) employed for diffuse runoff areas.

DRA	1976	1977	1978	Mean Coefficient
1	1.12 (0.06)	23.00 (1.23)	7.29 (0.39)	0.56
2	7.24 (0.13)	70.18 (1.26)	21.17 (0.38)	0.59
3	0.85 (0.05)	1.53 (0.09)	2.72 (0.16)	0.10
4	1.02 (0.12)	1.28 (0.15)	2.13 (0.25)	0.17
5	16.14 (0.16)	24.22 (0.24)	16.14 (0.16)	0.19
6	3.40 (0.34)	0.70 (0.07)	0.60 (0.06)	0.16
7	1.11 (0.10)	4.00 (0.36)	3.89 (0.35)	0.27
8	1.42 (0.03)	17.06 (0.36)	12.32 (0.26)	0.22
9	10.72 (0.05)	19.29 (0.09)	34.29 (0.16)	0.10
10	2.86 (0.12)	3.57 (0.15)	5.95 (0.25)	0.17
11	0.47 (0.12)	0.59 (0.15)	0.98 (0.25)	0.17
12	69.06 (0.10)	248.62 (0.36)	241.71 (0.35)	0.27
13	71.03 (0.10)	42.62 (0.06)	149.16 (0.21)	0.12
Total	186.44	456.66	498.35	

Table 36. Total nitrogen loadings (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1}) calculated for stream basins.

Stream	1976	1977	1978	Mean Coefficient
A	186.74 (0.22)	2168.85 (2.56)	1785.63 (2.10)	1.63
B	19.73 (0.32)	367.46 (5.95)	123.12 (1.99)	2.75
C	122.47 (0.46)	608.81 (2.27)	630.34 (2.35)	1.69
D	367.44 (0.56)	972.84 (1.47)	1277.29 (1.93)	1.32
E	4980.73 (0.67)	6686.06 (0.91)	24556.61 (3.32)	1.63
F	2637.23 (0.81)	7836.13 (2.40)	12575.29 (3.85)	2.35
G	74.03 -	84.85 -	93.41 -	-
H	105.00 (1.57)	44.17 (0.66)	77.86 (1.17)	1.13
I	311.42 (0.87)	556.04 (1.56)	498.98 (1.40)	1.28
J	69.48 (0.62)	536.96 (4.82)	155.58 (1.40)	2.28
K	221.36 (0.90)	233.67 (0.95)	141.52 (0.57)	0.81
L	1669.54 (0.25)	8036.86 (1.21)	21987.97 (3.32)	1.59
M	21.75 (0.35)	104.51 (1.67)	156.12 (2.50)	1.51
N	228.16 (0.67)	758.74 (1.77)	934.99 (2.18)	1.54
O	5134.55 -	28346.14 -	41467.47 -	-
Total A-M	10786.92	28237.21	64059.72	

Table 37. Total nitrogen loadings (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1}) employed for diffuse runoff areas.

DRA	1976	1977	1978	Mean Coefficient
1	5.05 (0.27)	79.66 (4.26)	38.34 (2.05)	2.19
2	21.72 (0.39)	228.93 (4.11)	120.87 (2.17)	2.22
3	9.52 (0.56)	24.99 (1.47)	32.81 (1.93)	1.32
4	6.89 (0.81)	20.40 (2.40)	32.73 (3.85)	2.35
5	87.78 (0.87)	157.40 (1.56)	141.26 (1.40)	1.28
6	15.70 (1.57)	6.60 (0.66)	11.70 (1.17)	1.13
7	3.89 (0.35)	18.54 (1.67)	27.75 (2.50)	1.51
8	10.43 (0.22)	121.34 (2.56)	99.54 (2.10)	1.63
9	120.01 (0.56)	315.02 (1.47)	413.60 (1.93)	1.32
10	19.28 (0.81)	57.12 (2.40)	91.63 (3.85)	2.35
11	3.16 (0.81)	9.36 (2.40)	15.02 (3.85)	2.35
12	241.71 (0.35)	1153.30 (1.67)	1726.50 (2.50)	1.51
13	419.08 (0.59)	561.14 (0.79)	2059.87 (2.90)	1.43
Total	964.22	2753.80	4811.62	

Considerable year to year and basin to basin variation exists in the quantities of nutrients that were exported through the Baptiste Lake watershed. In general, total annual nutrient yields (all stream inputs combined) followed the annual hydrological regime closely (Table 9). The annual discharge during 1978 was approximately two and a half times greater than that during 1977, and with the exception of nitrate-nitrite and ammonia all nutrient loads increased in a similar proportion; biotic activity and physical conditions within the streams may have played a more important role than stream discharge in determining the export of the latter two nutrients (Likens et al. 1977; Feller and Kimmins 1979; Kaushik et al. 1975; Kaushik and Robinson 1976).

Many streams were insignificant sources of nutrients to the lake, with several discharging for only a few weeks during and after spring runoff. The total input for streams "B", "C", "H", "I", "J", "K", and "M" was only a small fraction of the total input for all streams (e.g. 4% of total phosphorus and 3% of total nitrogen stream loadings during 1978). In the absence of extremely high nutrient concentrations, these ephemeral streams provide little useful data regarding nutrient loadings *per se*, although they can be used in a limited way to help illustrate the effects of land use on stream nutrient concentrations (see below).

Stream basins "E", "F", and "L", which are predominantly forested and together comprise 90 percent of the total watershed area, contribute a very high proportion of the total annual input of water and nutrients. As a result, the annual nutrient export/water discharge pattern for the complete Baptiste Lake watershed is tied closely to the relationship between these two variables for the forested basins.

The percent distribution of surface total phosphorus and nitrogen loadings into Baptiste Lake and through the outlet from 1976 to 1978 are presented in Table 38. The distribution of total organic nitrogen, inorganic nitrogen forms and SRP loadings are given in the Appendix in Tables A47 to A49. The south basin consistently received higher loadings of all nutrients than did the north basin and once again this was comparable to the breakdown of hydraulic loadings (Table 10). However, the south basin of the lake did not exhibit a higher level of phytoplankton production, and this probably reflects the influence of water column depth and stability on sedimentation processes and nutrient replenishment; the comparison between inputs of total nitrogen and total phosphorus with losses through the outlet also indicates the capability of the lake to trap non-conservative materials. For instance, during 1977 and 1978 approximately 60% of estimated inputs of total phosphorus from surface runoff alone were retained in the lake.

The seasonal fluctuations in daily total phosphorus and total nitrogen loadings to the lake from all streams combined are summarized in Tables 39 and 40. A large proportion of the annual nutrient input via streams occurs in conjunction with peak discharges during spring runoff (March and April). In addition, highly significant nutrient loads were generated as a result of heavy summer and fall storm events (August, 1976; June, 1977;

Table 38. Percent distribution of nutrient loadings through Baptiste Lake (kg yr^{-1})

Parameter	Total Input	South Basin	%	North Basin	%	Outlet
1976 Total-P	1895.03	1525.39	80.5	3369.64	19.5	391.52
Total-N	11751.14	9203.44	78.3	2547.70	21.7	5134.55
1977 Total-P	2732.78	1835.19	65.7	957.59	34.3	1036.88
Total-N	30991.01	20193.92	65.2	10797.09	34.8	28346.14
1978 Total-P	5124.69	3258.50	63.6	1866.19	36.4	2122.39
Total-N	68871.34	43243.43	62.8	25627.91	37.2	41467.47

Table 39. Monthly breakdown of total phosphorus loadings (kg mon^{-1}) via streams to Baptiste Lake (1976-1979).

Month	1976-77	%	1977-78	%	1978-79	%
Mar	-	-	313	15.0	327	7.3
Apr	1232	61.4	603	29.0	411	9.1
May	170	8.5	293	14.1	254	5.7
Jun	54	2.7	392	18.9	277	6.2
Jul	46	2.3	213	10.2	328	7.3
Aug	288	14.4	86	4.1	360	8.0
Sep	128	6.4	74	3.6	1582	35.2
Oct	60	3.0	81	3.9	840	18.7
Nov	27	1.3	23	1.1	115	2.6
Dec	0	0	1	0.05	0	0
Jan	0	0	0	0	0	0
Feb	0	0	0	0	0	0
Total	2005	100	2079	100	4494	100

Table 40. Monthly breakdown of total nitrogen loadings (kg mon^{-1}) via streams to Baptiste Lake (1976-1979).

Month	1976-77	%	1977-78	%	1978-79	%
Mar	-	-	1924	7.3	1773	2.8
Apr	6468	53.6	5397	20.3	4724	7.5
May	2293	19.0	4362	16.5	4139	6.6
Jun	504	4.2	7189	27.2	4131	6.6
Jul	431	3.6	3475	13.1	3038	4.8
Aug	1204	10.0	1150	4.3	4981	8.0
Sep	498	4.1	1152	4.4	24815	39.6
Oct	375	3.1	1348	5.1	13594	21.7
Nov	278	2.3	443	1.7	1364	2.2
Dec	6	0.05	17	0.06	79	0.1
Jan	6	0.05	9	0.03	8	0.01
Feb	6	0.05	7	0.03	7	0.01
Total	12069	100	26473	100	62653	100

September and October, 1978). This would suggest the need for flexible, event-based sampling since approximately one half of the 1978 stream total phosphorus loadings occurred as a result of extended rainfall periods during September and October. During winter (mid-November to mid-March), only stream "G" contributed any loadings of nutrients, and this was negligible in comparison to the annual total.

Throughout most of the year, the monthly percentage of the annual total phosphorus and total nitrogen loadings followed the discharge pattern closely (Table 11). The major deviation from this pattern occurred during the spring runoff when nutrient concentrations were elevated with respect to discharge. The relationships between nutrient concentrations and discharge are described in detail below.

8.2 OBSERVED EXPORT COEFFICIENTS

The annual total phosphorus and total nitrogen export coefficients developed for each tributary basin and diffuse runoff area are presented with the corresponding loading data in Tables 34 to 37; the data for the remaining forms of nitrogen and phosphorus are presented in the Appendix in Tables A25 - A28. As with the loading observations, a high degree of variability was observed between years and between basins for many nutrients.

The coefficients of variation calculated from nutrient export coefficients for all basins in each land use type in all years (n = 12) are presented in Table 41. For phosphorus, the least variation was found in stable (forested) basins. However, the high

Table 41. Coefficients of variation (%) for all stream basin export coefficients (1976-1978).

	TP	SRP	TN	TON	NH_3	NO_3+NO_2
Forested	58.9	66.3	78.9	84.0	53.6	86.3
Agricultural	127.0	153.1	83.7	78.8	111.4	126.3

Table 42. Coefficients of variation (%) for all export coefficients excluding ephemeral streams and 1976 nitrogen data.

	TP	SRP	TN	TON	NH_3	NO_3+NO_2
Forested	57.5	67.3	48.6	53.1	39.4	77.0
Agricultural	46.9	52.5	17.1	20.9	94.6	124.0

Table 43. Average nutrient export coefficients (kg ha^{-1} yr^{-1}) with modified coefficients in brackets.

	Agricultural Land	Forested Land
Total Phosphorus	0.43(0.27)	0.13(0.14)
Orthophosphate (SRP)	0.30(0.16)	0.05(0.05)
Total Nitrogen	1.86(2.13)	1.60(2.50)
Total Organic Nitrogen	1.39(1.60)	1.44(2.28)
Nitrate-N + Nitrite-N	0.32(0.26)	0.05(0.06)
Ammonia-N	0.16(0.13)	0.09(0.11)

coefficients of variation for both total and organic nitrogen were similar for forested and agricultural basins. Inorganic nutrient forms exported from agricultural watersheds displayed the greatest variations.

The high variability for both loadings and export coefficients can be partly explained on the basis of technique changes and on the inclusion of ephemeral streams into the data set. Most water samples collected from the streams during 1976 and a few from early 1977 were frozen due to logistical constraints in the laboratory, and this has introduced an unknown bias to the results. This is principally reflected in the nitrogen chemistry, as the concentration/loadings of total and organic nitrogen in 1976 were far less than those observed during 1977 and 1978 when compared with the corresponding volume of discharge. Conversely, the total phosphorus loadings for 1976 did not reflect as great a variance. These observations are generally supported by the results of Philbert (1973) who found marked decreases in dissolved organic nitrogen concentrations, alterations in relative ammonia/nitrate -nitrite levels, but only slight changes in total phosphorus in frozen water samples from the Great Lakes. Sample preservation outlined in the A.P.H.A. (1975) indicate that freezing is an acceptable method for storage for total phosphorus but not for total nitrogen analyses.

We have, therefore, assumed that the 1976 loadings of total and organic nitrogen are not useful in describing the loading characteristics of individual streams or the watershed in general. The inclusion of these data in Table 41 have undoubtedly increased the observed variance.

The hydraulic behavior of the small, ephemeral streams (e.g. "B" and "K") was erratic when compared to that of the larger streams. As mentioned previously, the total annual discharge for all streams during 1978 was approximately two and a half times higher than during 1977 yet many of the smaller streams displayed higher discharges during 1977. One concedes that this observation may be prejudiced by sampling design and frequency, as well as by the inherent variability of these small watercourses. Without the use of control structures and the development of discharge rating curves it is impossible to capture all peak flow events in any stream. However, the installation of such facilities is impractical on very small streams. Consequently, the loading and export coefficient estimates included in this data set may be subject to greater variability than estimates from large streams alone.

For comparison, the same analysis was conducted omitting the 1976 total and organic nitrogen data, and without the ephemeral stream in each data set ("K" for forested basins and "B" for agricultural basins) (Table 42). The coefficient of variation is substantially reduced for total and organic nitrogen export from both land use types. The total phosphorus and SRP data set for agricultural basins (n = 9) also shows a much reduced variance. There was no significant change in ammonia and nitrate-nitrite coefficients when ephemeral streams were excluded.

Mean export coefficients were calculated for each nutrient by combining the average figures for the three years of study for each stream in each land use category. The coefficients presented in Table 43 include those from all streams for all years and in brackets show the modified figures excluding ephemeral streams ("B" and "K") and the 1976 total/organic nitrogen data.

A simple non-parametric test which estimates Spearman's correlation coefficient was used to determine the general relationship between land use and export of total phosphorus and nitrogen. All emphemeral stream data were included in this test. The results indicate that at a significance level of $P<.01$, the amount of total phosphorus lost per unit area increased as the percent composition of agricultural land increased within basins. For total nitrogen, however, it was concluded that the type of land use had little bearing on the export of this nutrient ($P>0.1$). Because loadings of total nitrogen were thought to be unusually low during 1976, mean coefficients calculated using only 1977 and 1978 data were tested against percent composition of land use in basins. The results, however, still indicated no significant difference in export of total nitrogen from forested or agricultural lands.

Upon close examination of the unmodified export coefficients one observes that the dominant component of the total nitrogen load for both land use types is organic nitrogen, and that the export coefficients for organic nitrogen are similar. The reasons for this similarity are unclear, although over 90 percent of the total nitrogen in the dark grey wooded soils found in the Baptiste Lake region occurs in the organic form (D Penny, personal communication).

The Mann-Whitney U-Test (Siegel, 1956) was performed on data from both land use types to determine if the apparent differences between nutrient export were statistically significant. For both modified and unmodified TP/SRP export coefficients there was a significant difference between agricultural and forested land use at $P < .05$. However, for the remaining nutrient forms no significant differences were detected.

The total phosphorus and total nitrogen export coefficients derived for the Baptiste watershed are compared to those reported by other workers in Table 44. The ranges, where available are shown in brackets.

Phosphorus export from both forested and agricultural lands in the Baptiste watershed is similar to that derived for sedimentary areas elsewhere, but would appear to be substantially higher than that from stream basins draining igneous bedrock areas characteristic of the Canadian Shield (Dillon and Kirchner 1975). Total nitrogen export, however, appears to be relatively low when compared with the limited data available from other sources; this seems to be the case particularly in agricultural basins. Only the nitrogen export coefficients of Rast and Lee (1978) modified for use in the western United States are as low as our figures. Since their data represent a consolidation of data from many other studies we recommend the tentative use of their export coefficients for calculating theoretical loadings to lakes in the boreal mixed-wood forest biome. We anticipate that the acquisition of more data from this region will verify this recommendation.

Table 44. Total phosphorus and total nitrogen export coefficients with ranges shown in brackets.

Author	TP (kg ha^{-1} yr^{-1})	TN (kg ha^{-1} yr^{-1})
Rast and Lee (1978)		
Forested	0.1	3.0 (1.0)[a]
Agricultural	0.5	5.0 (2.0)[a]
Sonzogni and Lee (1974)		
Forested	-	-
Agricultural	0.7	5.0
U.S. EPA (1974)		
Forested	0.1	4.0
Agricultural	0.3	10.0
Uttormark et al. (1974)		
Forested	0.2 (.05-.80)	2.5 (1.0-5.0)
Agricultural	0.3 (.01-1.00)	5.0 (2.0-10.0)
Dillon and Kirchner (1975)		
Forested		
-Igneous	.05 (.01-.09)	
-Sedimentary	.12 (.07-.18)	
Forested/Pasture		
- Igneous	.10 (.06-.16)	
- Sedimentary	.23 (.11-.37)	
This study		
Forested	.13 (.02-.25) .14[b]	2.07[c] (.57-3.85) (2.50)[b,c]
Agricultural	.43 (.03-2.1) .27[b]	2.59[c] (1.67-5.95) (2.13)[b,c]

a - coefficients suggested for use in the western U.S.
b - excludes ephemeral streams
c - excludes 1976 results

8.3 STREAM NUTRIENT CONCENTRATIONS

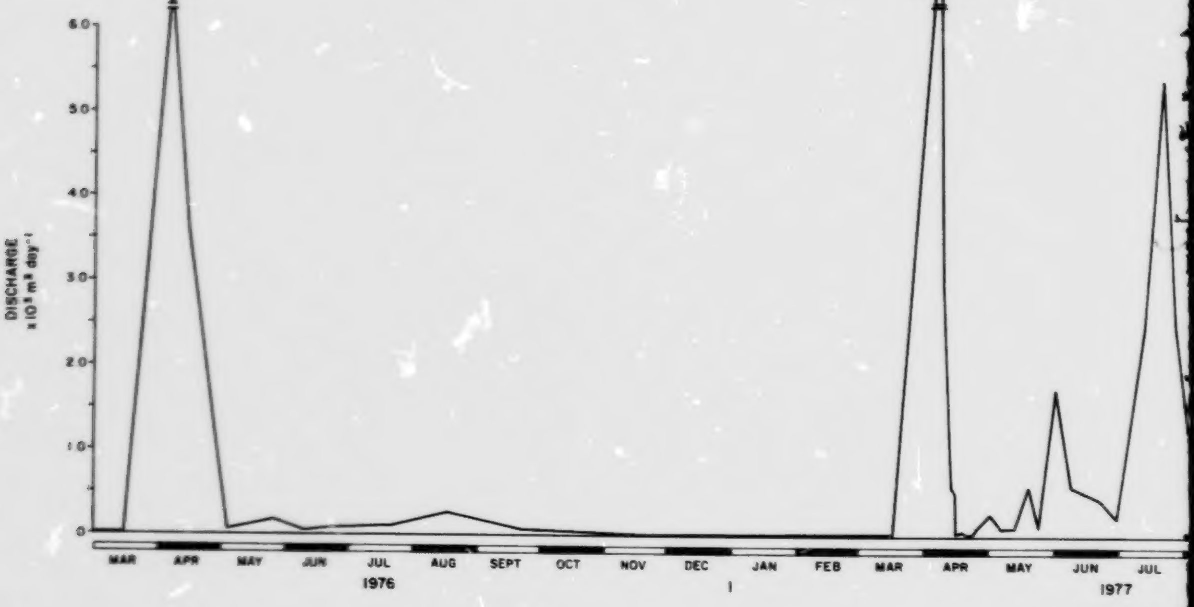
Variations in either streamflow or nutrient concentrations cause variations in nutrient export; therefore,it is therefore desirable to determine the relative significance of each component in order to accurately estimate or predict accurately the export of nutrients from specific watersheds. This type of information is also necessary in order to establish the frequency of sampling required to derive reliable estimates of nutrients exported from different land use types.

In the following sections, we will present the seasonal and annual concentration trends followed by a discussion of the relationship between discharge and nutrient concentrations in streams of agricultural watersheds, with emphasis on stream "A", and streams of forested watersheds, with emphasis on stream "E". The outflow of nutrients and water, and their relative importance in estimating nutrient export through the outlet stream will also be discussed.

8.3.1 AGRICULTURAL RUNOFF

Streams "A" and "N" were the only agricultural stream basins characterized by continuous discharge from spring to late autumn. Stream "B" and "M" flowed only in response to the spring snowmelt each year. During 1976 stream "N" ceased to flow after the first week in June.

Seasonal fluctuations in TP and SRP concentrations for stream "A" are illustrated in Figure 58. The variations in discharge for the same period are illustrated in Figure 57. The highest concentrations generally coincided with high discharges



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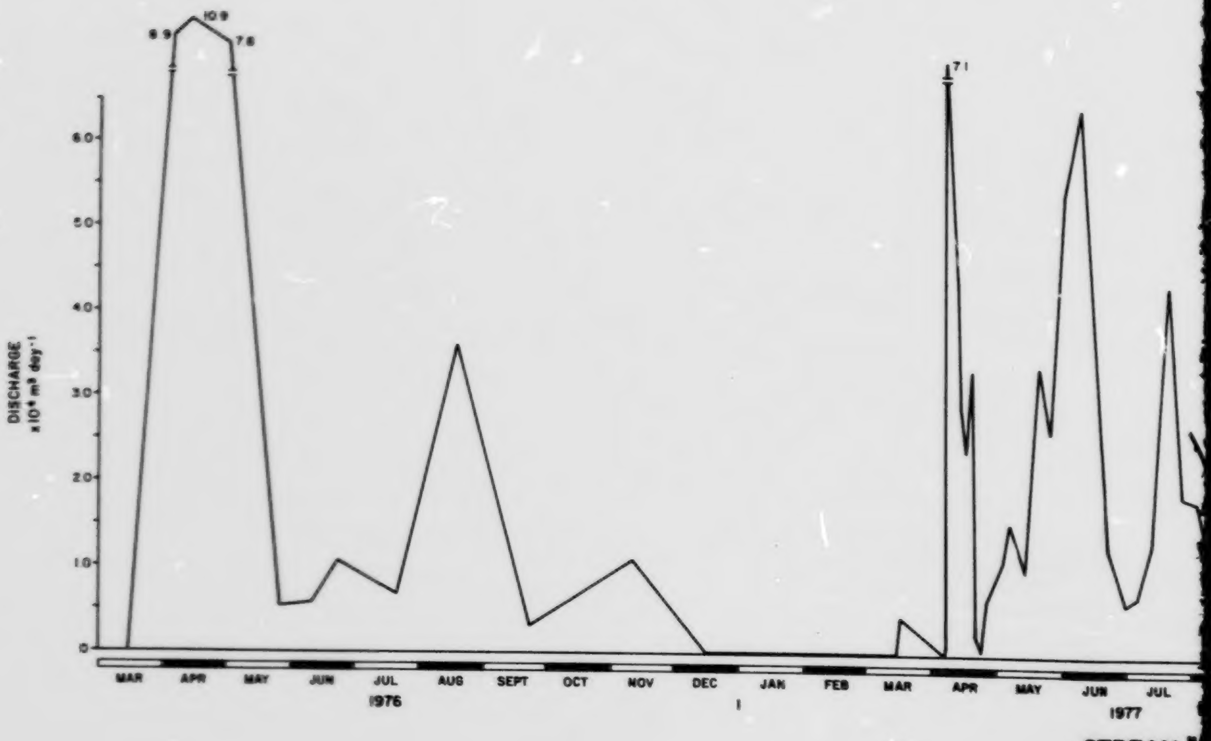

STREAM

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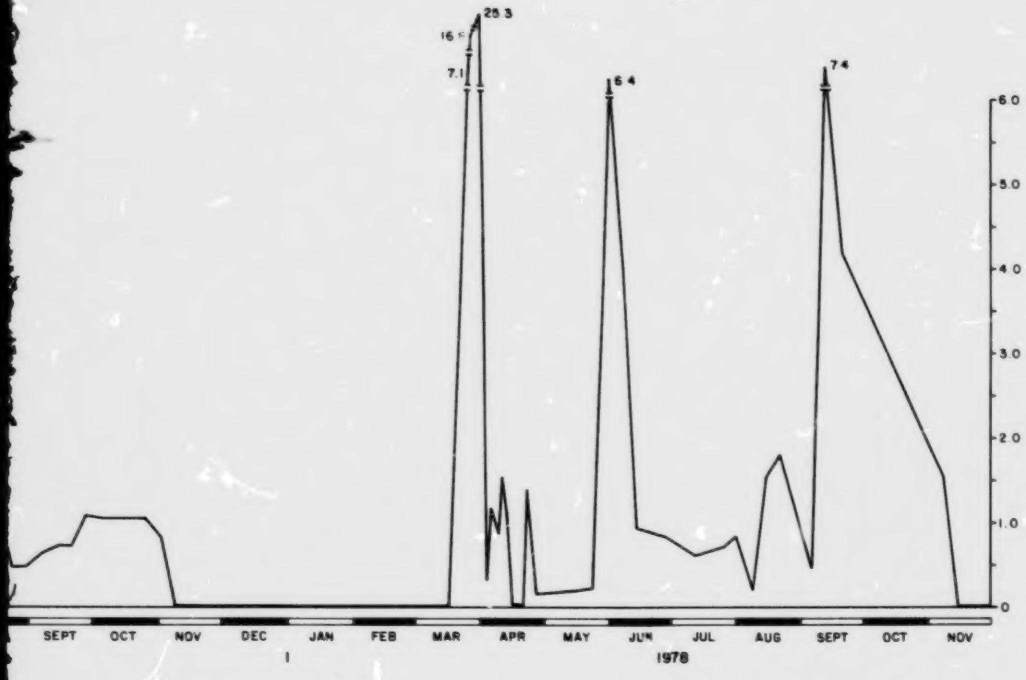


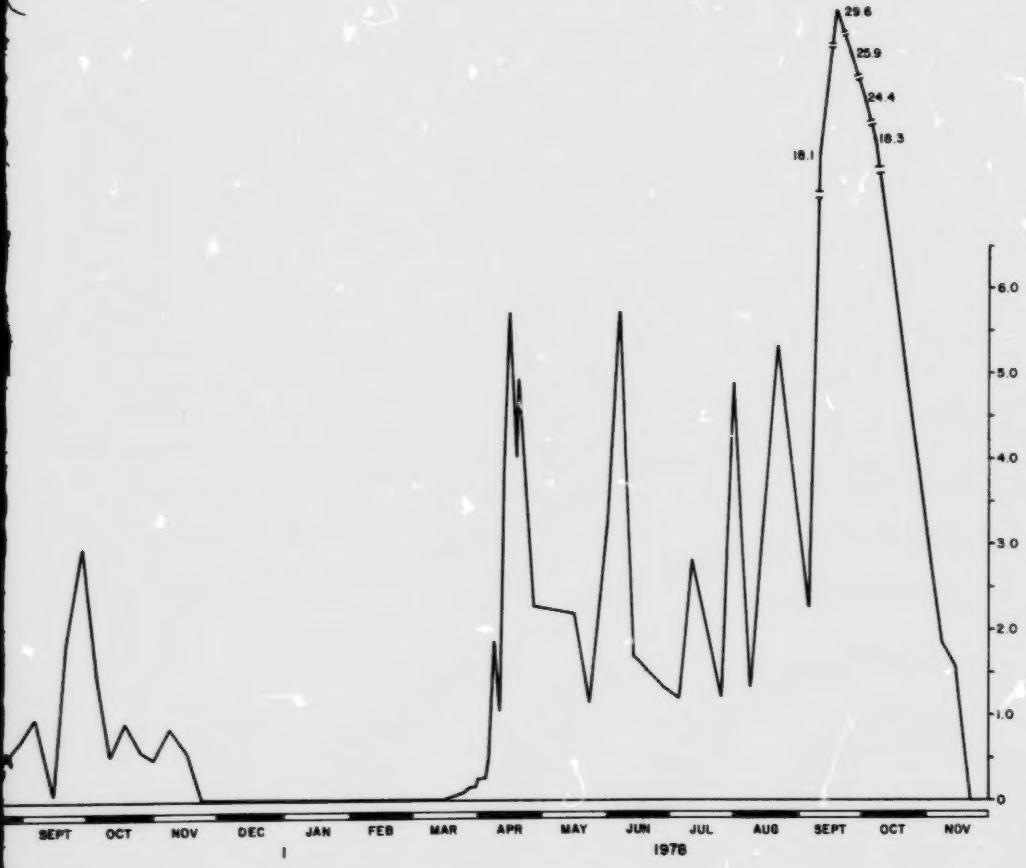
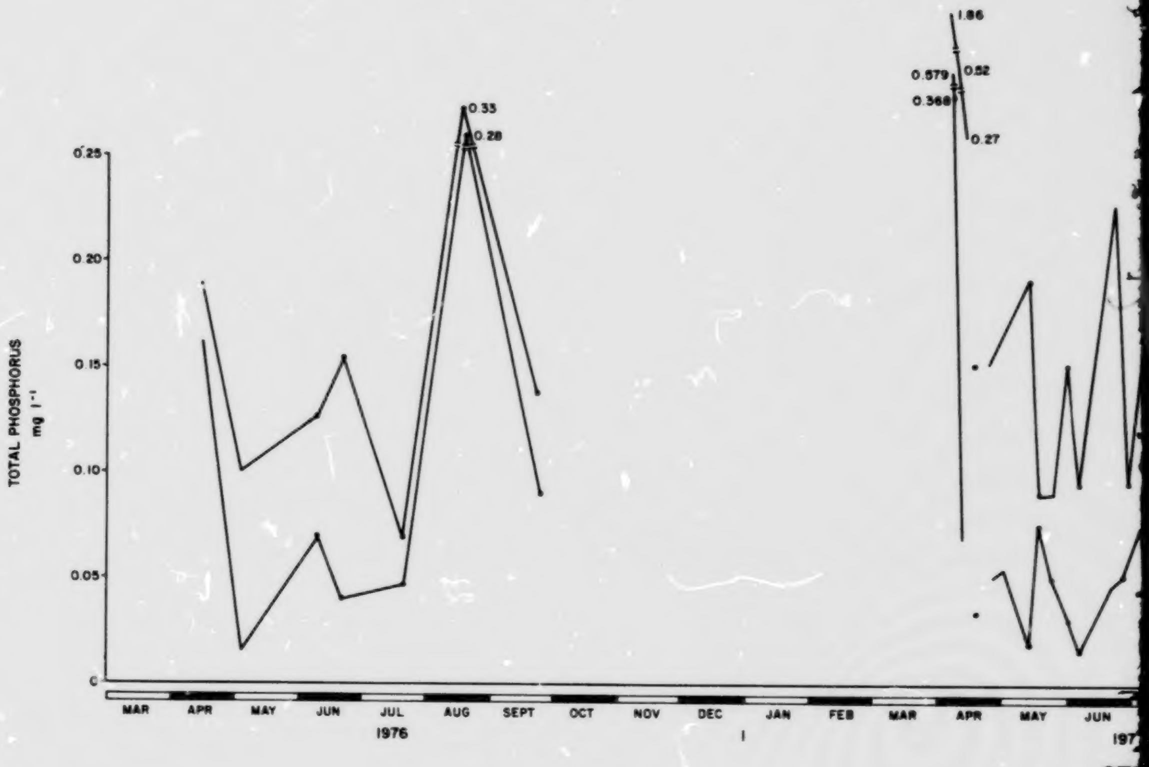
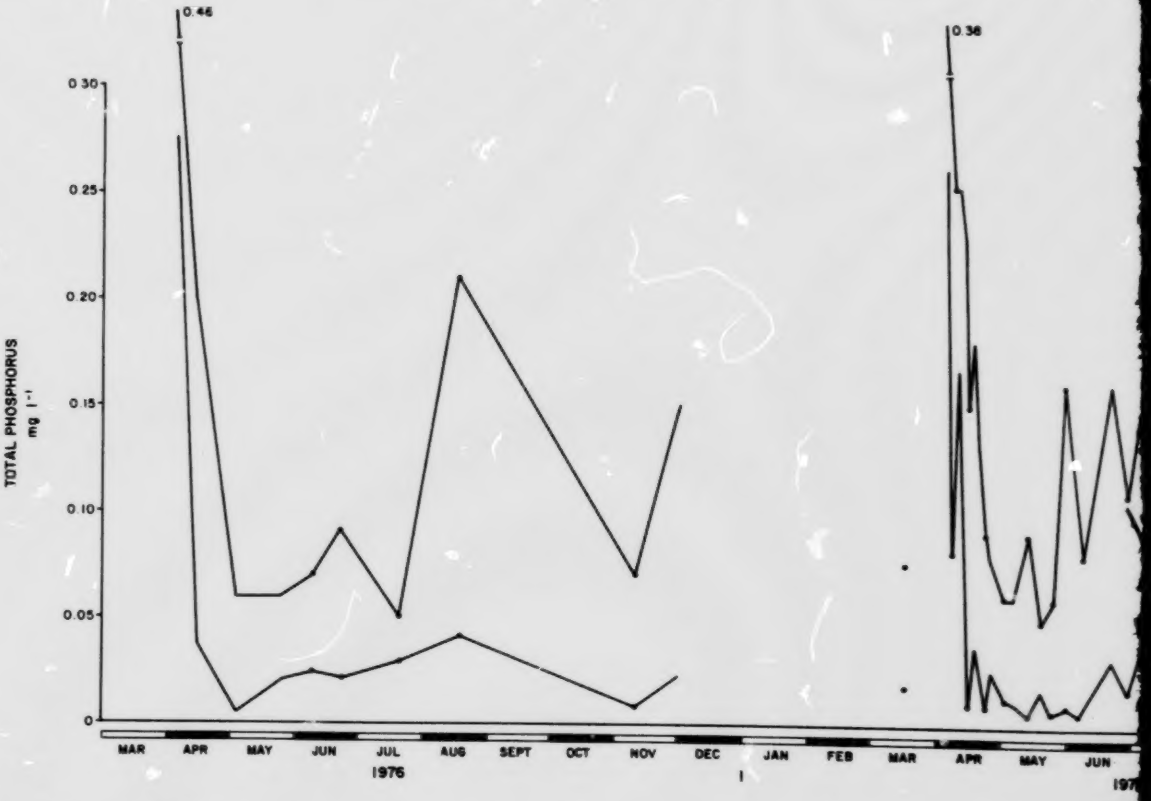


FIG.57 SEASONAL FLUCTUATIONS IN STREAM DISCHARGES



STI



STR

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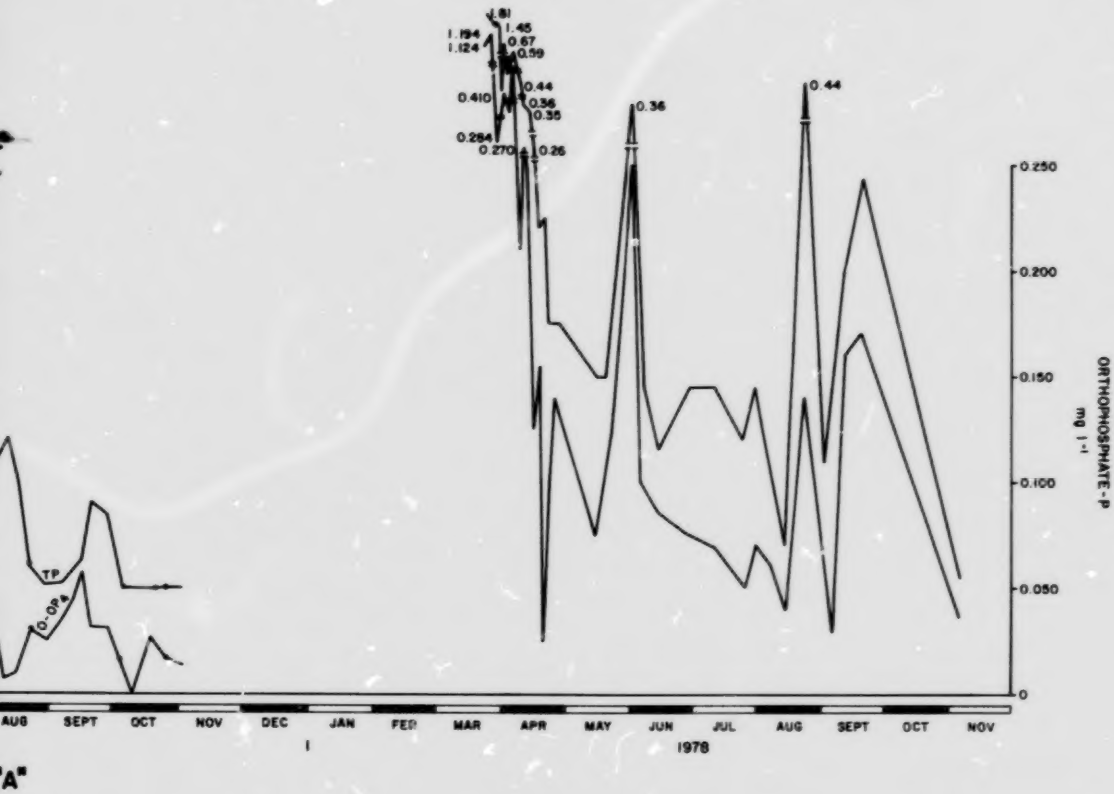


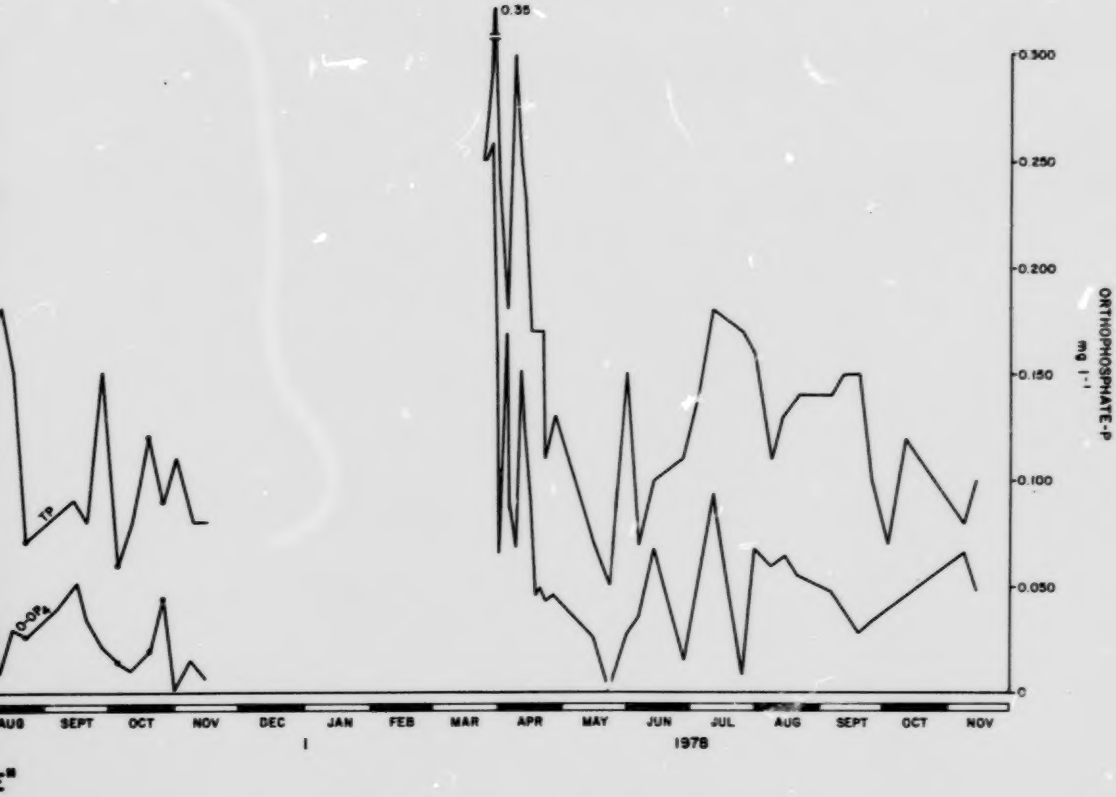


FIG.58 SEASONAL FLUCTUATIONS IN STREAM TOTAL PHOSPHORUS AND ORTHOPHOSPHATE (SRP) CONCENTRATIONS

(i.e. spring runoff and heavy precipitation events), and during these events, SRP formed a significant portion of the phosphorus pool. Mid-summer concentrations of both forms of phosphorus remained high relative to lakewater levels. In the other agricultural streams, total phosphorus concentrations and the portion of soluble reactive phosphorus were also the highest during spring runoff each year.

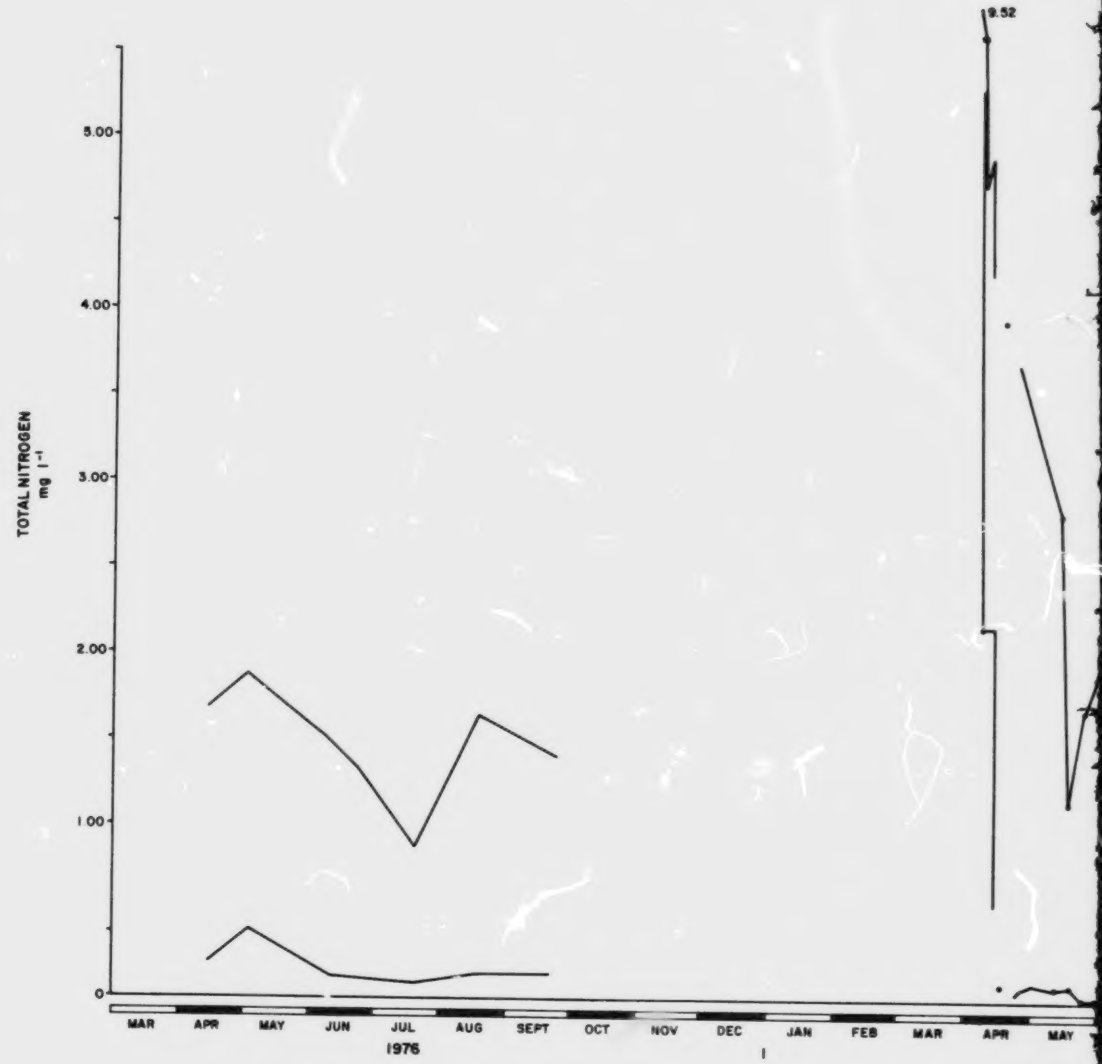
The seasonal fluctuations in total nitrogen, total organic nitrogen and total inorganic nitrogen for stream "A" are illustrated in Figures 59 and 60. Concentrations of total, total inorganic and total organic nitrogen displayed patterns which appeared relatively constant between 1977 and 1978. High concentrations were associated with spring runoff, after which there was a decline to concentrations which varied little in relation to discharge, except during occasional storm events. These latter peaks are apparent during July, 1977, May and August 1978, and probably reflect the short-term scouring of stream bed materials as a result of high flows.

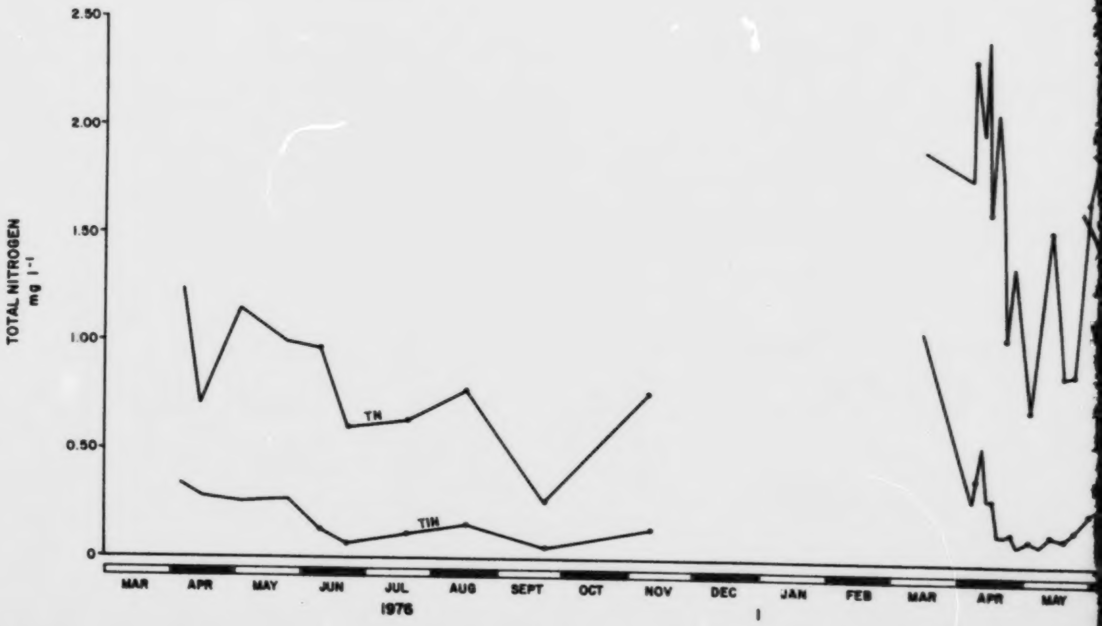
During early April of 1977, the total inorganic fraction made up the largest portion of the total nitrogen pool; a similar pattern was observed in April, 1978, although the inorganic values were slightly lower that year. Throughout the remainder of the year, organic nitrogen represented the dominant component. These patterns were similar in other agricultural streams.

The seasonal fluctuations in ammonia and nitrate-nitrite are illustrated in Figures 61 and 62, respectively. Generally, ammonia exhibited higher concentrations in 1977 than in 1978, although the reasons for this are difficult to determine without some indicator of microbial activity. The summer of 1978 was warmer and brighter than 1977 and this may have promoted more carbon fixation, as suggested by the higher mean pH observed during May - August 1978 ($\bar{x}$ = 7.9 n = 14) as opposed to that observed in 1977 ($\bar{x}$ = 7.4, n = 12). The combination of elevated oxygen concentrations and higher pH may have encouraged the chemical oxidation of ammonia, the physical evolution of the gas, and nitrification processes.

The spring runoff of 1978 was characterized by some extremely high ammonia concentrations during late March and early April. These were short-lived occurrences and were probably not observed during 1977 due to the less frequent sampling that year. The overwinter degradation of stream bed detritus with the concomitant oxygen depletion in reaches free from continuous water movement may have contributed to the high ammonia levels, which subsequently fell rapidly with declining discharges and increasing temperatures.

Nitrate-nitrite was also high during early spring. Nitrate is stored during winter in the snowpack and can be concentrated by evaporation; spring snowmelt can subsequently supply much of this nutrient to streams during runoff (Feller and Kimmins 1979). Summer concentrations were generally low, probably reflecting uptake by growing plant populations, and trends toward higher concentrations in the autumn may be due to the leaching and conversion of nitrogenous decomposition products from decaying vegetation (Feth 1961). The major peaks in nitrate-nitrite appear to lag those of ammonia by several weeks.


TOTAL NITROGEN
mg l^{-1}
5.00
4.00
3.00
2.00
1.00
0
9.52
MAR
APR
MAY
JUN
JUL
AUG
SEPT
OCT
NOV
DEC
JAN
FEB
MAR
APR
MAY
1976


TOTAL NITROGEN
mg l^{-1}
2.50
2.00
1.50
1.00
0.50
0
TN
TIN
MAR
APR
MAY
JUN
JUL
AUG
SEPT
OCT
NOV
DEC
JAN
FEB
MAR
APR
MAY
1976

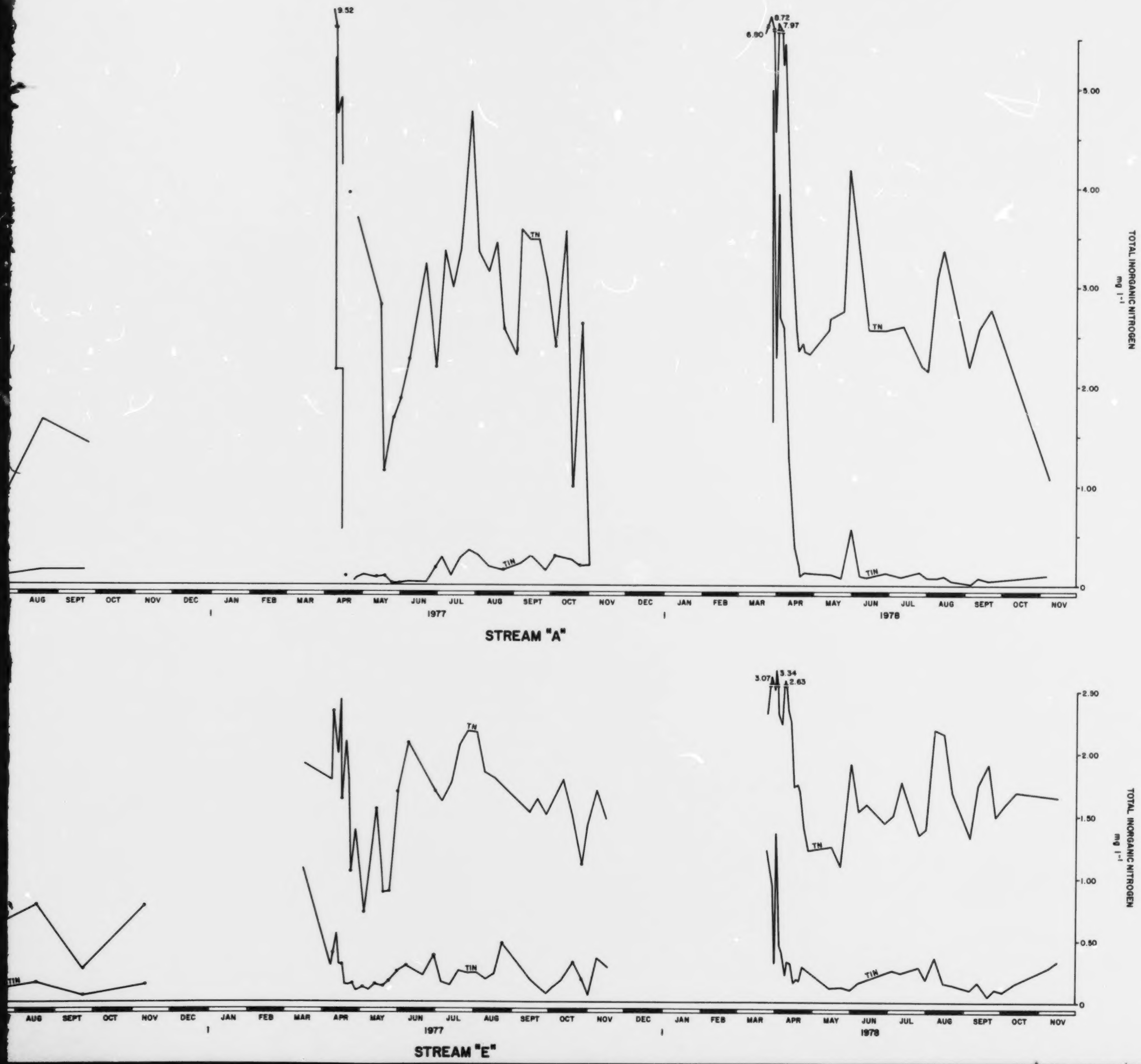


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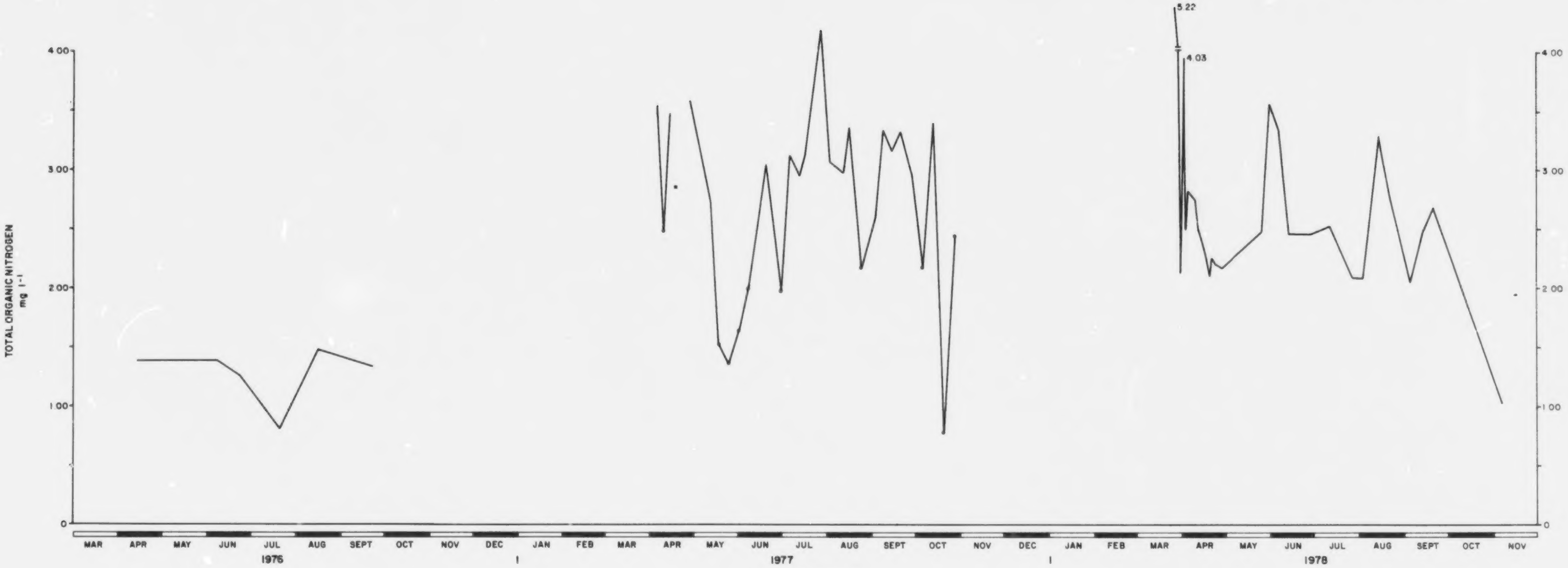
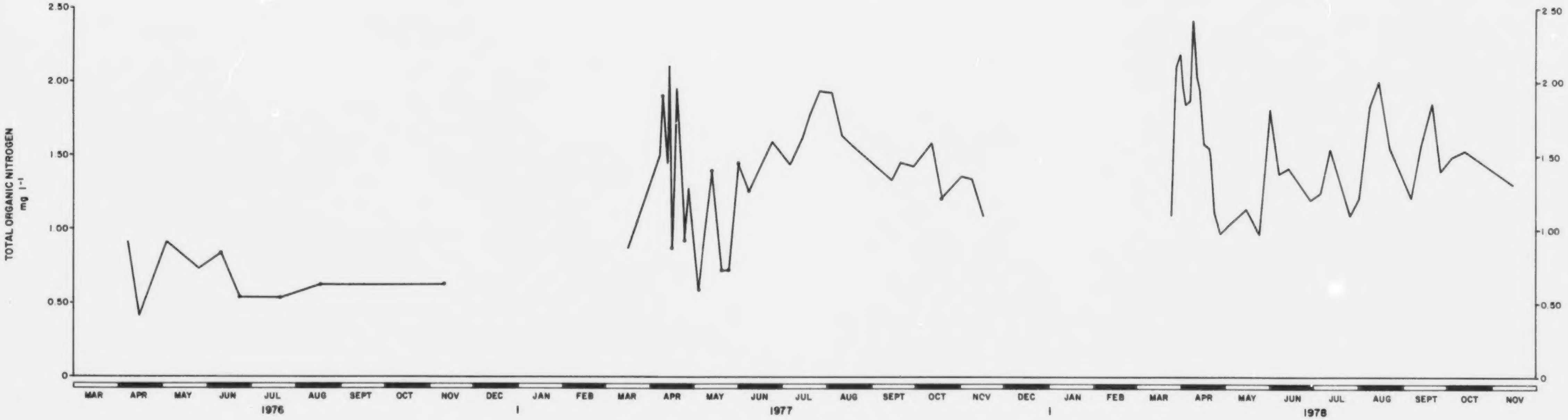
FIG. 59 SEASONAL FLUCTUATIONS IN STREAM TOTAL NITROGEN AND TOTAL INORGANIC NITROGEN CONCENTRATIONS



STREAM "A"



STREAM "E"

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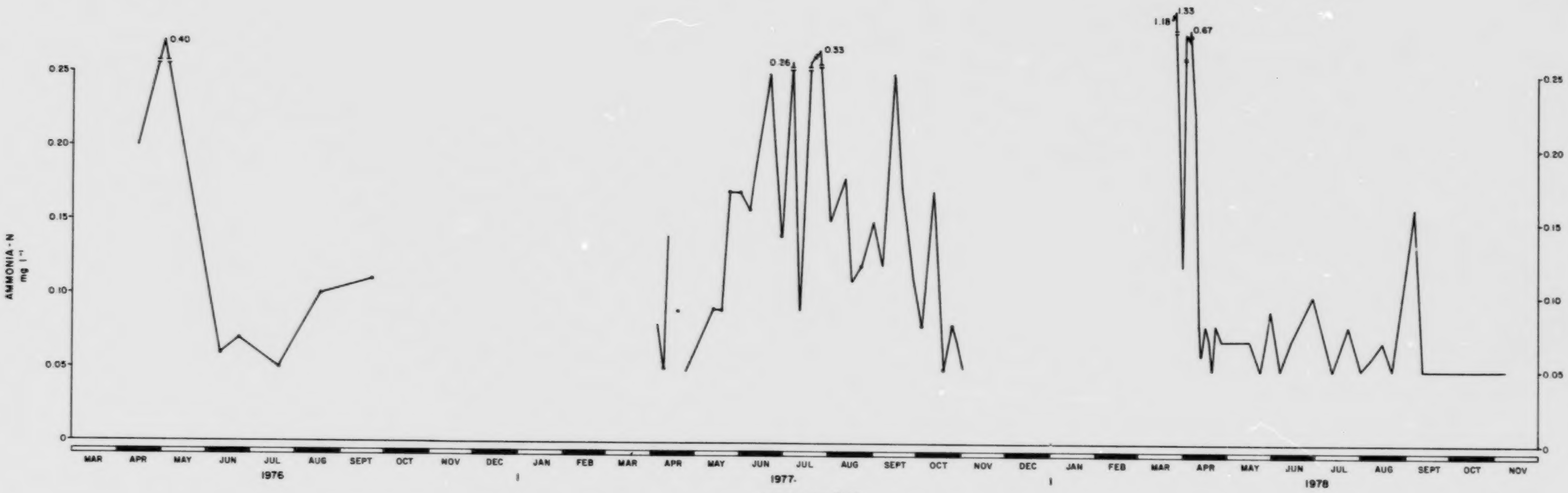
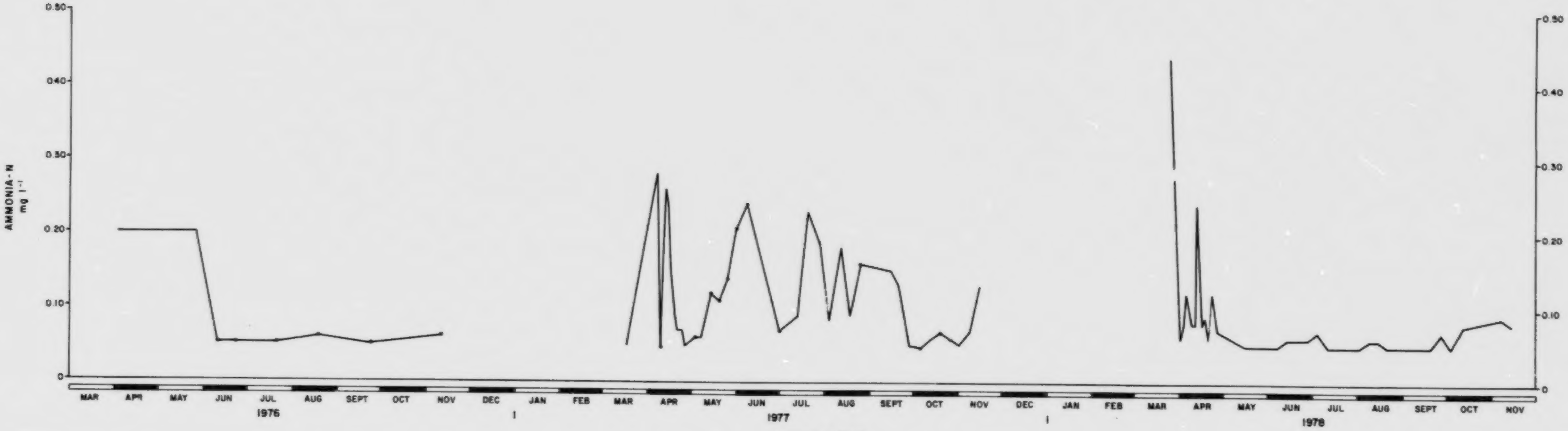
FIG. 60 SEASONAL FLUCTUATIONS IN STREAM TOTAL ORGANIC NITROGEN CONCENTRATIONS



STREAM "A"



STREAM "E"

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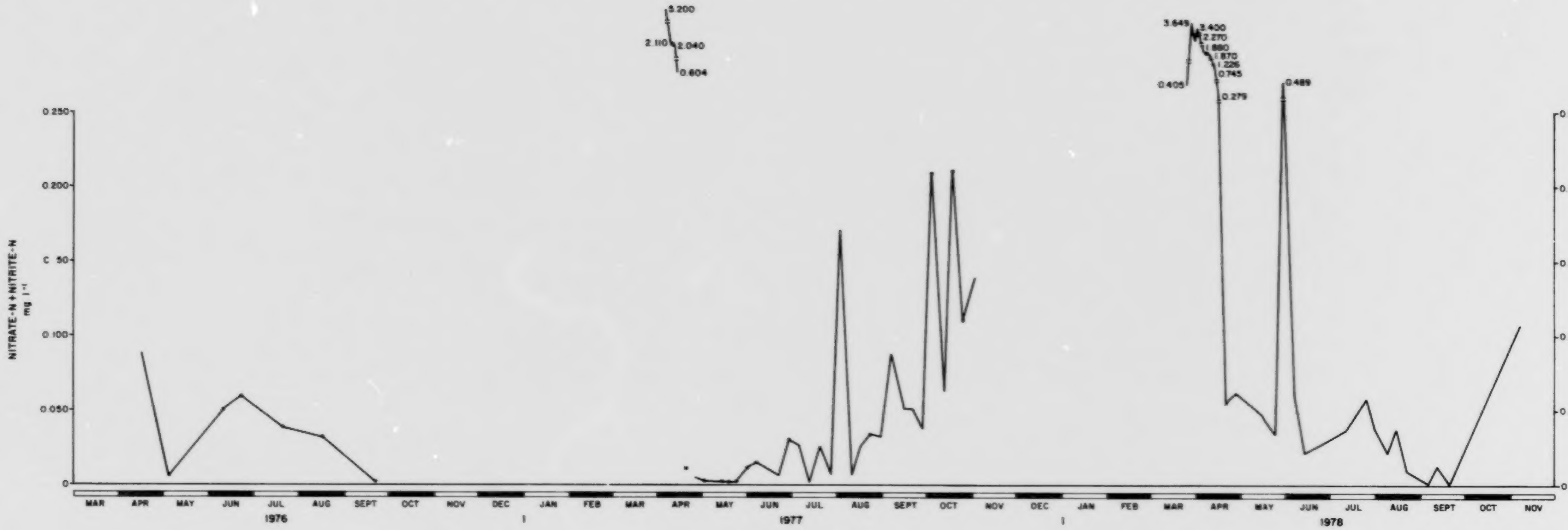
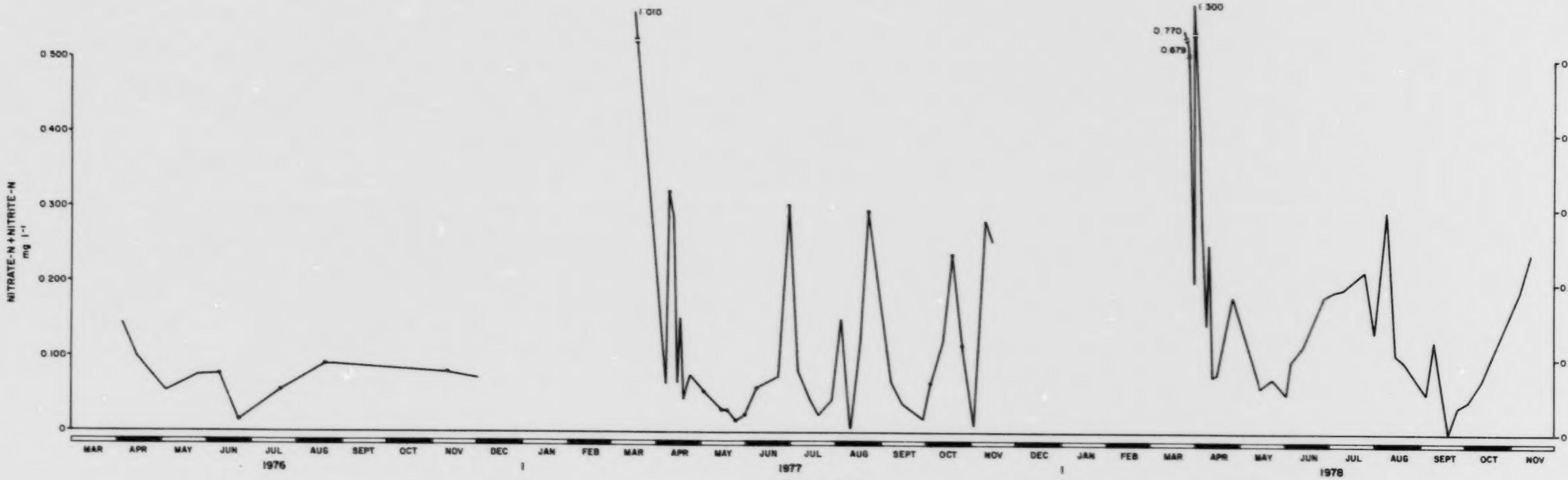
FIG. 61 SEASONAL FLUCTUATIONS IN STREAM AMMONIA CONCENTRATIONS



STREAM "A"



STREAM "E"

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FIG. 62 SEASONAL FLUCTUATIONS IN STREAM NITRATE AND NITRITE CONCENTRATIONS

8.3.2 FORESTED BASIN RUNOFF

The seasonal variations in TP and SRP concentrations for stream "E" are illustrated in Figure 58. A sharp increase in concentrations of both nutrients was associated with spring runoff during each year, although 1978 concentrations peaked before discharge. Local roadside drainage entering stream "E" at our sampling site, or snow melt adjacent to the stream which contained high amounts of road dust may have resulted in these high concentrations at such low discharge.

After spring, concentrations remained relatively stable in relation to the change in discharge, although a pronounced minimum was evident in May of both 1977 and 1978. There did not appear to be consistent increases in concentrations during any of the summer or fall storm events with the exception of peaks in total phosphorus on August 17, 1976, September 27, 1977 and July 11, 1978 in response to peaks in discharge. The range in concentrations was fairly similar between years. In general, the loss of both forms of phosphorus appears to be comparatively stable from year to year in this forested stream basin, in spite of significant variations in discharge. However, many investigations of nutrient dynamics during storm events have indicated a pronounced hysteresis effect, and it is quite possible that our sampling regime may not have accurately captured peaks in stream nutrient concentrations during storms. This latter type of sampling requires a much more sophisticated effort than was the case during the Baptiste Lake study.

The seasonal fluctuation in total, organic, and total inorganic nitrogen for stream "E" are illustrated in Figures 59 and

60; the variation in stream discharge is shown in Figure 57. Once again, samples which had been frozen displayed lower concentrations than what was expected; otherwise, the seasonal patterns were similar to those observed in the agricultural streams. During 1977 and 1978, spring peaks for total, organic, and total inorganic nitrogen were followed by lower concentrations which, except during storm events, remained relatively constant relative to discharge for the remainder of the open water season. The bulk of daily total nitrogen concentrations consisted of organic nitrogen except two dates during spring runoff (March 16, 1977 and March 24, 1978) when total inorganic nitrogen concentrations were slightly higher.

The seasonal fluctuations of ammonia and nitrate-nitrite concentrations for stream "E" are illustrated in Figures 61 and 62 respectively. The pattern for this stream was similar to stream "A", with the highest concentrations of the year being associated with spring runoff. After spring runoff in 1977, concentrations of both nutrients fell rapidly and displayed occasional peaks throughout the remainder of the growing season. For ammonia, these high concentrations were associated with high streamflow and high organic nitrogen during mid-June and mid-July. In the case of nitrate-nitrite, a lag of a few weeks behind ammonia peaks is also apparent. As observed in stream "A", the 1977 ammonia concentrations are higher than those measured in 1978, with the latter being frequently below the minimum detection limit (< 0.05 mg L^{-1}).

All other forested basin streams except "K" flowed continuously from spring to late autumn each year. In all streams, total and soluble reactive phosphorus concentrations were at their maximum during each spring. Concentrations decreased with the decline in flows and remained relatively constant thereafter, in spite of the large fluctuations in discharge. Similarly, total and organic nitrogen concentrations were always high during spring runoff in these streams; after this period concentrations decreased. It appeared that on May 31 and July 19, 1977, and on May 30, July 11, and August 13, 1978 in streams "F" and "L" that increases in discharge were accompanied by increases in both total and organic nitrogen concentrations. These increases, however, were small in comparison to the increases in discharge.

Ammonia and nitrate-nitrite concentrations in streams "F" and "L" behaved similarily to those in stream "E" and the agricultural streams. Concentrations were high during spring runoff and dropped as discharge decreased. Ammonia concentrations during the 1977 summer and autumn were higher and more variable than those in 1978. Ammonia peaks on July 7, 1977 in stream "F" and July 26, 1977 in "F" and "L" followed increases in discharge and organic nitrogen. These were followed, in approximately two weeks, by increases in nitrate-nitrite. Autumn ammonia levels were low, whereas nitrate-nitrite remained high until winter.

8.4 ANNUAL FLOW-WEIGHTED MEAN CONCENTRATIONS

Flow-weighted mean concentrations (annual loading divided by annual discharge) have been calculated for all nitrogen and phosphorus forms in streams of both land use types. The data for total phosphorus and total nitrogen are presented in Tables 45 and 46. The data for the remaining nutrient forms are presented in the Appendix in Tables A50 to A53. A three-year mean is presented for each stream for each nutrient. Total and organic nitrogen means excluding 1976 data are also given for each stream.

As observed with the loading figures and export coefficients, considerable year to year and basin to basin variation exist. The ephemeral streams ("B" and "K") once again stand out as being atypical from the larger streams of similar land use; total phosphorus means for both streams reflect much higher concentrations than observed elsewhere.

Considerable variation within years and between basins was generally observed for most nutrients in streams draining agricultural basins. Only total nitrogen (coefficient of variation = 24.5%) and organic nitrogen (coefficient of variation = 20.2%) were characterized by levels which were relatively similar between streams and between years for individual streams. Although ammonia levels for stream "N" did not appear to remain to remain constant from year to year based on flow-weighted mean results, ammonia levels in 1978 were generally much lower (usually below the detectable limit). The flow-weighted mean calculation can be influenced by high loadings which may occur over short time intervals, consequently an unusually high ammonia concentration on March 29, 1978 of 6.94 mg L^{-1} may have resulted in an unrepresentative value for that year.

A similar observation was made on stream "A" during 1977 and 1978. The 1978 flow-weighted mean was much higher than that

Table 45. Annual flow-weighted mean concentrations (mg L^{-1}) for total nitrogen

Land Use	Stream	1976	1977	1978	$\bar{x}$	$\bar{x}$ Excluding 1976
Agric.	A	1.56	5.52	3.66	3.58	4.59
	B	1.88	5.19	4.16	3.74	4.68
	M	1.71	3.07	5.05	3.28	4.06
	N	1.51	2.83	3.63	2.66	3.23
Forested	E	0.87	1.66	1.66	1.40	1.66
	F	1.02	2.06	2.03	1.70	2.05
	K	1.56	3.89	2.85	2.77	3.37
	L	1.25	2.40	2.70	2.11	2.55

Table 46. Annual flow-weighted mean concentrations (mg L^{-1}) for total phosphorus

	Stream	1976	1977	1978	$\bar{x}$
Agri.	A	0.18	0.78	0.46	0.47
	B	0.49	1.83	1.06	1.13
	M	0.47	0.66	0.70	0.61
	N	0.86	0.39	0.62	0.62
For:	E	0.15	0.12	0.12	0.13
	F	0.15	0.13	0.13	0.14
	K	0.25	0.61	0.12	0.33
	L	0.15	0.14	0.18	0.16

during 1977, whereas concentrations were generally higher in 1977. This reversal may have resulted either from missing peak concentrations during 1977 spring sampling period (which have a major influence on the calculation), or from the presence of unusually high concentrations measured in the 1978 spring runoff period (March 24/78 = 1.18 mg L^{-1}; March 27/78 = 1.33 mg L^{-1}, Figure 61).

Unlike the results for agricultural basin streams, there is generally less year to year and/or basin to basin variation in flow-weighted mean concentrations for the forested basin streams. Total phosphorus means were found to vary the least between basins, as well as from year to year (coefficient of variation = 13.5%). Soluble reactive phosphorus means were not as similar from basin to basin (coefficient of variation = 43.2%) as they were between years for individual streams (coefficient of variation; E = 16.2%, F = 33.3 %, and L = 22.4%).

Total and organic nitrogen levels were relatively stable from year to year for individual stream as well as between basins for both years (coefficient of variation; total nitrogen = 19.7%, organic nitrogen 22.6%). Stream E daily concentrations were lower than those for streams "F" and "L", and therefore, resulted in slightly lower flow-weighted means. Conversely, ammonia and nitrate-nitrite means were highly variable both between basins and between years for individual streams.

A statistically significant difference (Mann-Whitney U-test; $P < .05$) was found between total nitrogen flow-weighted (modified) means from agricultural and forest drainage. This is in contrast to our analysis of export coefficients in which no differences could be detected. This may be attributed to the difference in areal water yields between the two basin types, wherein the average yield for forested basins was approximately 56 percent higher (573 m^3 ha^{-1}) than that of agricultural basins (380 m^3 ha^{-1}), resulting in similar export per unit area. The same test was applied to the other nutrient forms with the result that significantly higher concentrations ($P<.05$) were always found in agricultural runoff.

The SRP/TP flow-weighted mean concentration ratio was determined for both land use types excluding ephemeral streams. For agricultural land there was a much higher proportion of SRP in the phosphorus pool (58% as opposed to 33% from forested land). A similar ratio for inorganic/total nitrogen indicates that approximately 25% of the runoff agricultural land could be in a readily available form, whereas only 13% would be immediately available in forest runoff.

This comparison combined with the export coefficient data suggests that while agricultural land exports approximately two to three times as much total phosphorus as does forested land (Table 43), the available proportion of the total also increases resulting in an approximate four to six-fold increase in the potentially available loading.

8.5 N:P MASS RATIOS

The seasonal fluctuations in total N:P mass ratios for all tributary streams combined are illustrated in Figure 63. Due to the questionable nitrogen results in 1976 the ratios during that year may have been higher than indicated. With the exception of slightly lower total N:P ratios during spring runoff (March and April) no definite annual pattern was observed, and ratios generally ranged between 10 and 25. Omernik (1977) found a typical ratio for mostly forested watersheds was 25.

Figure 64 illustrates total N:P mass ratios for each year for individual streams. Disregarding 1976 bars, forested basins were characterized by higher ratios than basins with a high proportion of agricultural land. This resulted from less phosphorus being lost from forested lands since total nitrogen export was almost equal from the two basin types. The ratios in the outlet are higher than inputs each year, and are probably due to the acquistion of nitrogen from other sources during the passage of water through the lake coupled with the loss of phosphorus through sedimentation. Input 'G' (groundwater) ratios are high due to the insignificant loading of phosphorus.

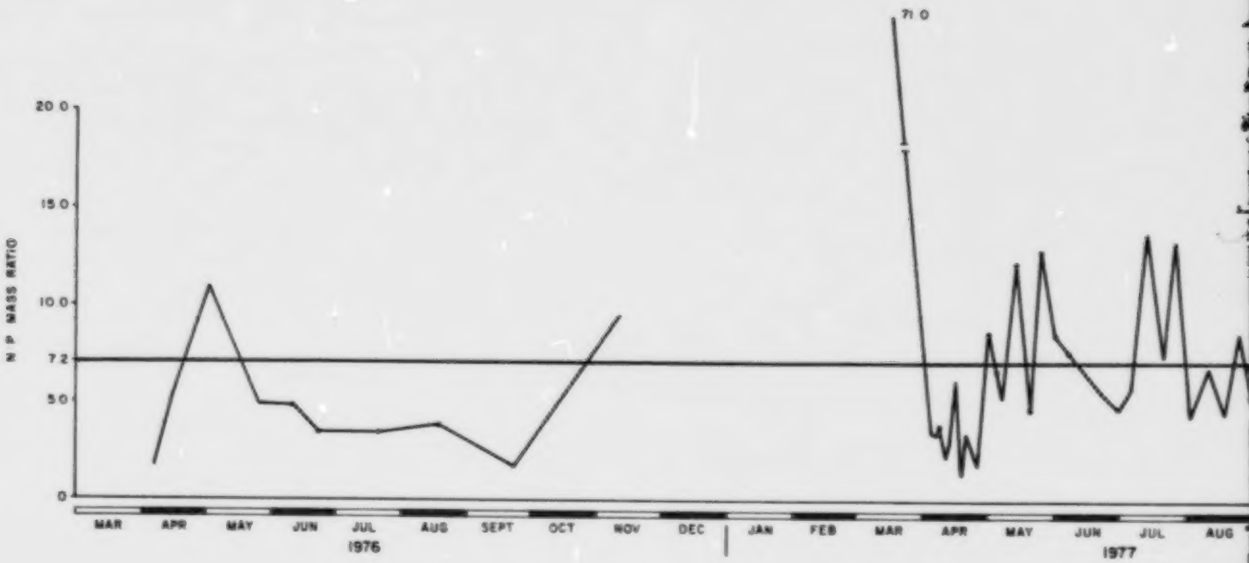
When inorganic forms of nitrogen and phosphorus were used to determine "available" N:P ratios different patterns emerged. N:P ratios during 1977 peaked irregularly in March, May, July and November (Figure 63). Ratios during other months were generally below 7.2. All ratios during 1978 were below 7.2 and as noted earlier, this suggests that surface water inputs were nitrogen deficient.

In order to explain the differences between 1977 and 1978 ratios, we compared total annual loadings of total inorganic nitrogen (ammonia plus nitrate-nitrite) to the annual soluble reactive phosphorus loadings. Inorganic nitrogen loadings (all inputs combined) increased by only 19 percent from 1977 to 1978 whereas soluble reactive phosphorus increased by 136 percent.

Streams "E", "F", and "L" have the most influence on the inorganic N:P mass ratio estimates, since they contribute to a large proportion of the daily inorganic nitrogen and phosphorus loading. During 1977 and 1978 these streams comprised 57 and 82 percent respectively of the total inorganic nitrogen input and 46 and 83 percent respectively respectively of the SRP total input, and while there was only a 71 percent increase in total inorganic nitrogen from 1977 to 1978, soluble reactive phosphorus increased by 325 percent in these streams. The increases in annual SRP loadings for these basins may be attributed to the annual increase in hydraulic loads from 1977 to 1978. A similar trend was not apparent for inorganic nitrogen; this may be due to factors other than stream discharge having control over inorganic nitrogen concentrations.

When the total inorganic nitrogen loading was divided by that for soluble reactive phosphorus for individual streams, ratios during 1977 were generally higher than those in 1978 (Figure 65). However, all were less than or close to 7.2 with the exception of those for input "G" and the 1977 ratio for the outlet stream. The outlet discharge appears to change from phosphorus deficiency in 1977 to nitrogen deficiency in 1978. This results from a 35 percent decrease in total inorganic nitrogen outflow loading and a 31 percent increase in soluble reactive phosphorus outflow from 1977 to 1978.

One concludes that the surface water inputs to Baptiste Lake are low in "available" nitrogen in relation to "available"



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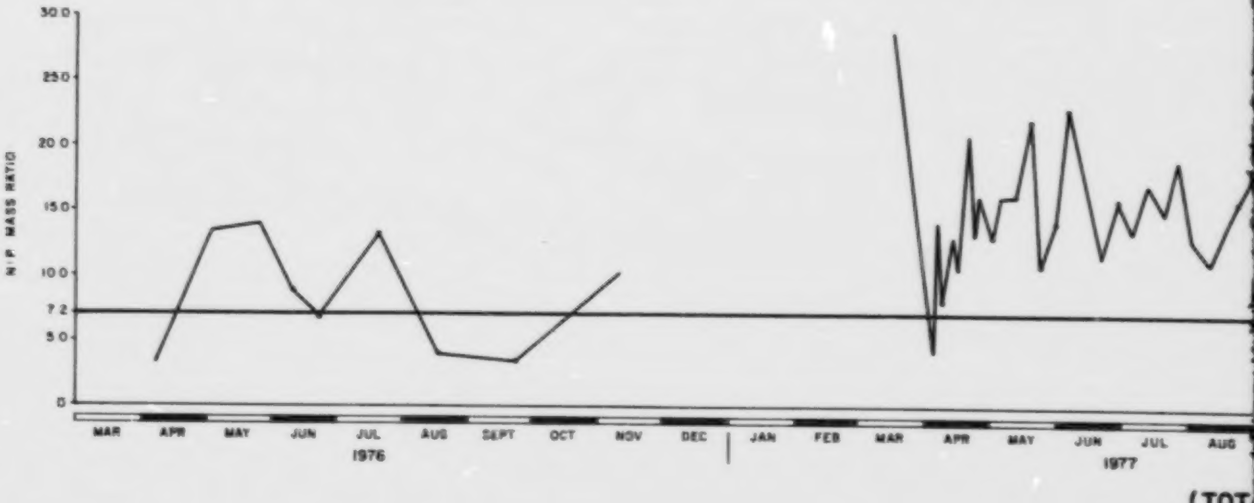

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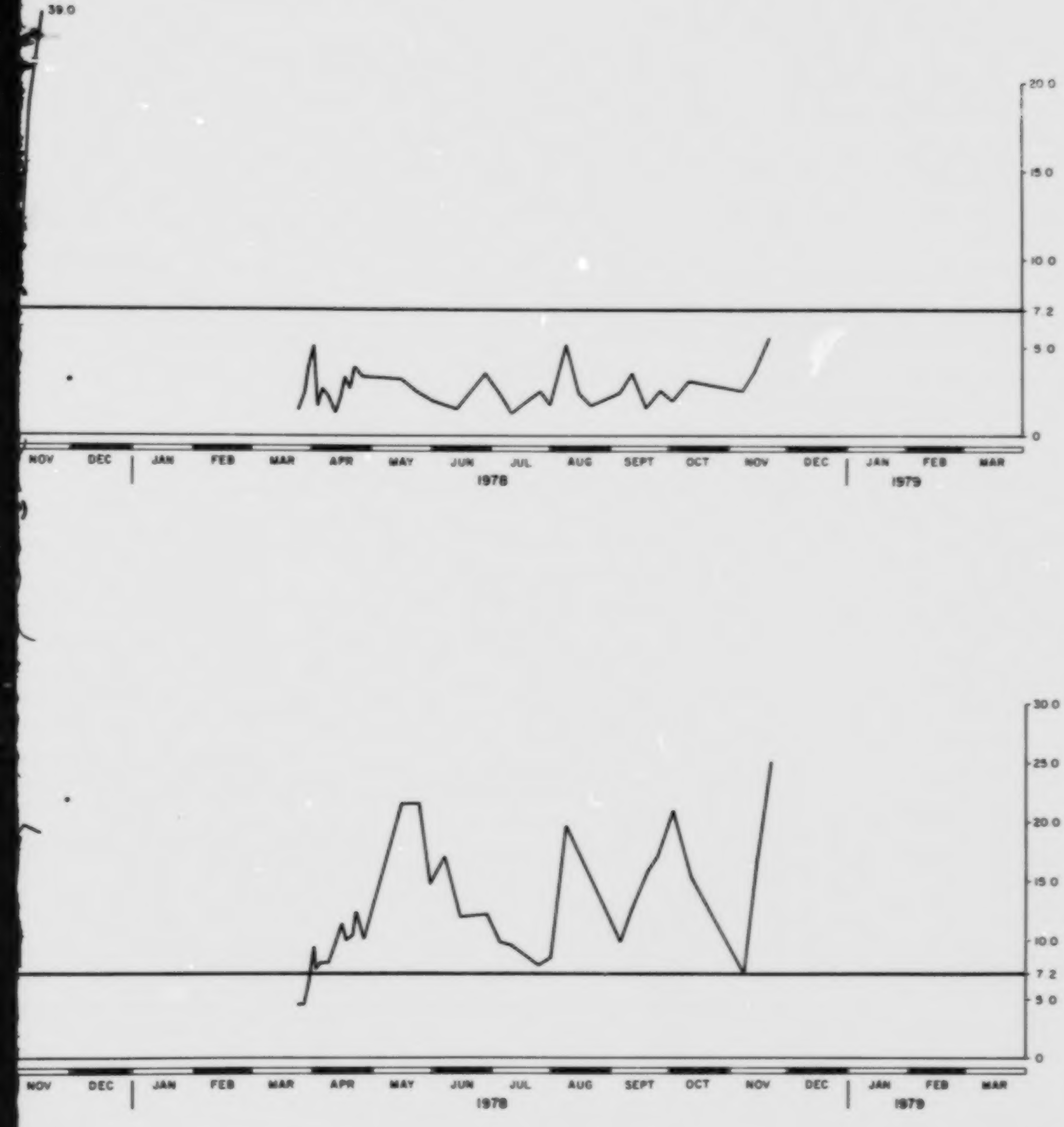

FIG. 63 SEASONAL FLUCTUATIONS IN N:P MASS RATIOS FOR ALL STREAM INPUTS INTO BAPTISTE LAKE

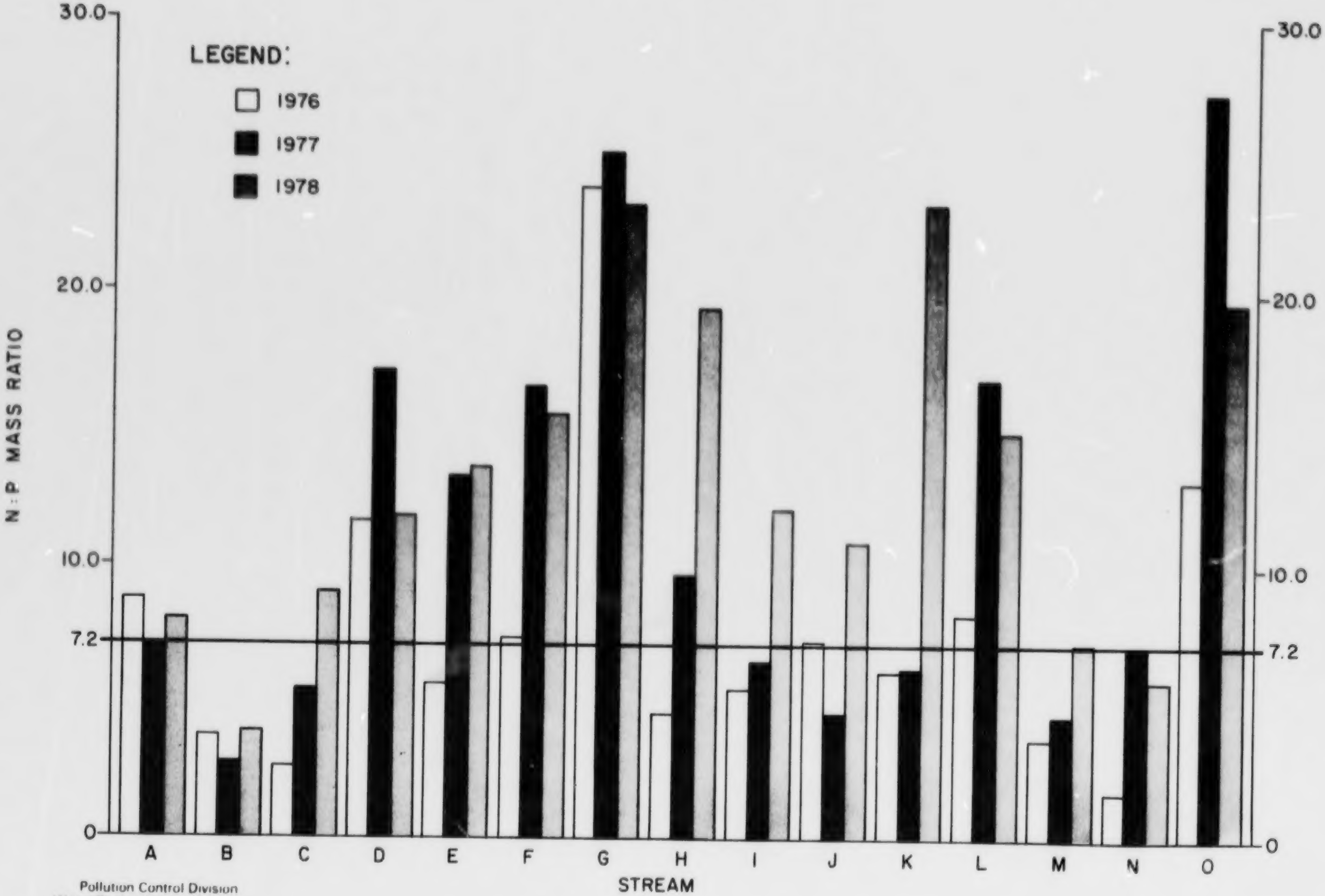

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FIG. 64 ANNUAL N:P MASS RATIOS (TOTAL FORMS) FOR ALL STREAM INPUTS INTO BAPTISTE LAKE

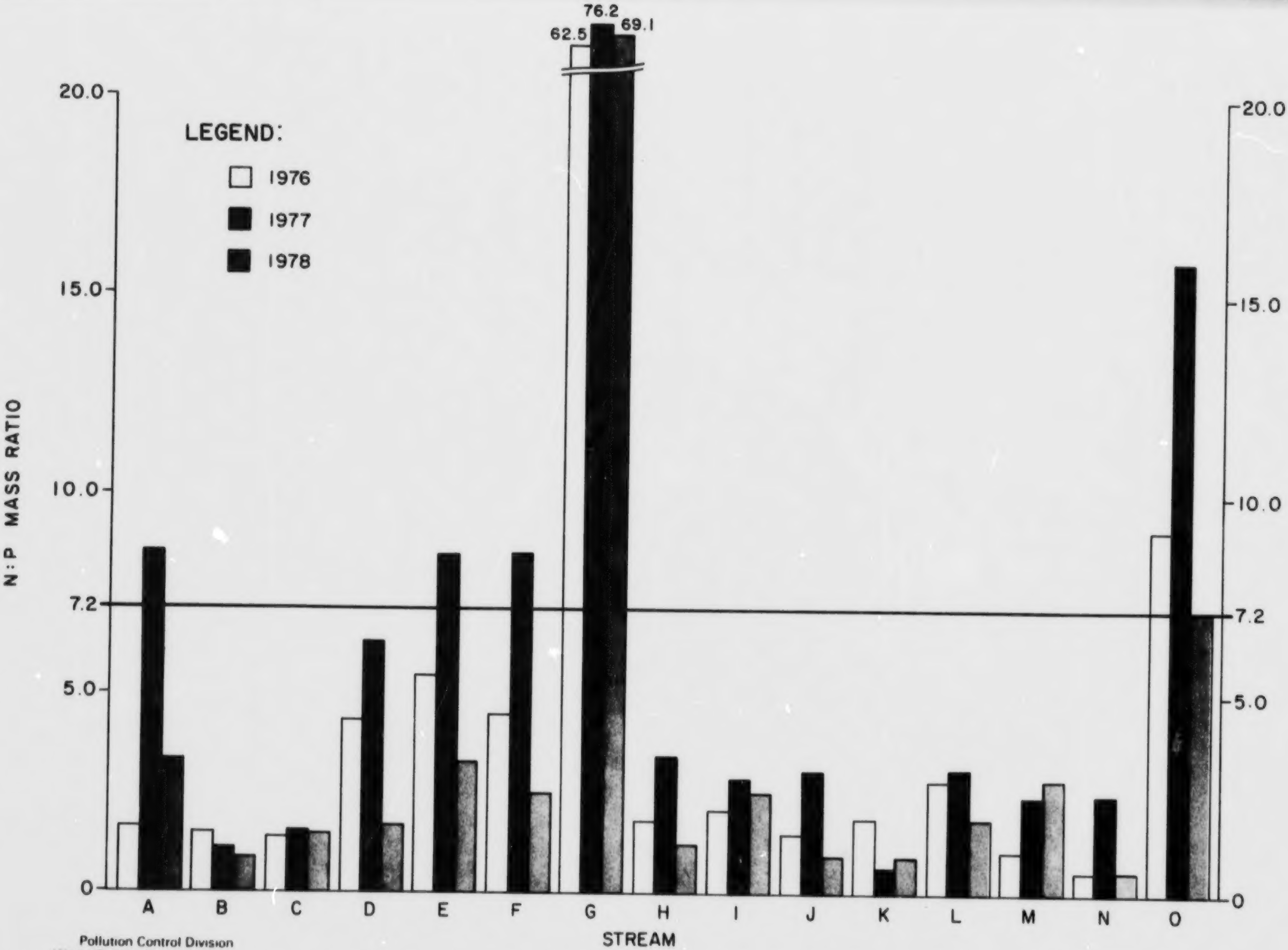

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FIG. 65 ANNUAL N:P MASS RATIOS (AVAILABLE FORMS) FOR ALL STREAM INPUTS INTO BAPTISTE LAKE

phosphorus. This is no doubt a major determinant of inorganic nitrogen levels in the lake and therefore may contribute to the success of those members of the phytoplankton community capable of nitrogen fixation.

8.6 NUTRIENT CONCENTRATION AND DISCHARGE RELATIONSHIP

It is important to estimate accurately nutrient export through surface runoff for this can often be the largest source of incoming nutrients to a lake. Although the best export estimate involves obtaining continuous measurements of discharge and frequent determination of nutrient concentrations, it is not often attempted, nor was it in this study. However, if discrete nutrient concentrations can be related to discrete discharge measurements over a wide range of flows, then concentrations may be interpolated from the more commonly available continuous stream discharge data. This would allow for a more reliable export estimate to be calculated.

In order to assess the variability of nutrient concentrations with discharge, regression analyses were run on discrete concentrations of the six nitrogen and phosphorus species against discrete discharge values for streams of forested and agricultural basins. Logarithmic transformation was used to normalize discharge and chemical concentration data.

During periods of stable flow, nutrient concentrations are highly influenced by biological processes. This activity is distinctly seasonal and in order to determine any relationships between concentration and discharge we have attempted to minimize of biologically-mediated changes by dividing the water year into three seasons. The criteria used to define these seasons were temperature, discharge, and various chemical ratios (total organic carbon : total organic nitrogen; ammonia : nitrate-nitrite; and ammonia : total kjeldahl nitrogen).

Spring conditions were assumed to occur from March 15 to April 30, summer from May 1 to October 10, and autumn from October 11 to when streamflow ceased at the beginning of winter (generally near the end of November). However, our analyses revealed little in the way of useful predictive relationships. Generally, we found that if a nutrient did appear to vary with discharge (either directly or indirectly) the regression was never strong. While it was observed in both agricultural and forested basin streams that several peaks in discharge during the summer and autumn of 1977 and 1978 were accompanied by peaks in total and organic nitrogen, the corresponding regression slopes were neglible, indicating little variation in the concentrations over a wide range of discharges.

In addition, we found no statistically significant relationship between ammonia or nitrate+nitrite concentration with discharge in any of these streams. We conclude that levels of these nutrients in Baptiste streams are probably more dependent on instream biological processes than discharge rates.

Limitations in this analysis arose from the use of data derived from infrequent sampling and, therefore, we were not able to assess adequately the variability of concentration with short-term changes in streamflow for streams of either land use type. The analyses were additionally complicated by not sampling and testing for the relationships that may exist with either particulate or total dissolved nutrient forms and discharge.

8.7 NUTRIENT LOADING AND DISCHARGE RELATIONSHIP

As previously discussed, the annual export of nutrients is strongly dependent upon the annual discharge for any given stream basin. Furthermore, the hydrological regime has been shown to be highly variable for all of the Baptiste Lake tributaries during the three years of observation.

Mean export coefficients of the type generated in this study and in others are usually based upon several years of observations. Consequently, the year to year hydrological fluctuations are built into the coefficients, and their best applications are obviously over the longer term; their validity to any short-term budgeting problem is therefore questionable. However, we have shown that an alternate expression of the annual loading, the flow-wieghted mean, displays a much reduced year to year variability for certain nutrients; i.e. more independence from annual discharge. These observations suggest a simple alternative for calculating nutrient loadings for streams where these data are available.

Specifically, this would involve combining the established, average flow-weighted mean for a given stream with any subsequent measurements of total annual discharge for the same watercourse. According to data presented previously, this technique could be used to derive annual loadings for total phosphorus, soluble reactive phosphorus, total nitrogen, and organic nitrogen from the large forested basins of the Baptiste Lake watershed ("E", "F", and "L") since the year to year variability of these flow-weighted means are low. Moreover, the basin to basin variabilities are also low which suggest an overall average figure could be generally applied to the forested lands of this watershed.

However, the applicability of this approach to agricultural stream basins is limited by the high variabilities associated with the flow-weighted means of most nutrients; it would appear that the annual export of nutrients via agricultural runoff is not easily predictable unless long-term records are available.

To illustrate this approach, the measured total phosphorus loadings (Lp; kg yr^{-1}) for these forested stream basins ("E", "F", and "L") were regressed against measured annual discharge (Q; m^{-3} yr^{-1}) for the same streams for the three study years. A strong relationship was found:

$$L_p = 0.13Q + 68.59 \quad (n = 9, r^2 = .93)$$

Although this type of regression has been criticized as being spurious (Benson 1965) since one is effectively plotting "xy" against "x", it does serve to emphasize the low variability of the TP flow-weighted means over time and space, and the fact that a reasonable degree of confidence may be associated with loadings derived in this way. A logical extension of this observation would be the ability to predict the loadings expected during extreme hydrological conditions (e.g. the 1:25 high and low runoff years) and also for the long-term average situation.

8.8 GENERAL COMPARISON BETWEEN AGRICUTURAL AND FORESTED BASINS

The quality of water flowing from undisturbed forested watersheds is generally regarded as high, and is invariably affected by clearing. Feller and Kimmins (1979), and Likens et al. (1977) have indicated that removal of forest vegetation as well as disruption of agricultural lands accelerates erosion and increases the loss of particulate and dissolved materials from the watershed. As a result, chemical concentrations in stream water usually increase and may vary markedly over time and space.

Our data generally support these observations. For instance, the annual concentration range for total phosphorus, soluble reactive phosphorus, total nitrogen and organic nitrogen appear relatively constant from year to year within forested basins. Furthermore, the total phosphorus flow-weighted mean concentrations are similar between forested basins.

However, discharge from agricultural streams exhibited considerably more year to year and basin to basin variabilities in the levels of nutrients. Flow-weighted mean concentrations were generally higher than those observed in the forested stream basins.

The total phosphorus and total nitrogen export coefficients developed for these land use types in the Baptiste Lake watershed fall within the range presented by others. The loss of total phosphorus increases as the percentage of agricultural land increases; in contrast, land use appeared to have little bearing on the loss of total nitrogen. Moreover, statistically significant differences between the export of phosphorus from agricultural lands as opposed to forested lands have been determined.

Since we have verified these differences the long-term impact of historical land clearing activities on the phosphorus loading regime can be estimated. The theoretical loading estimate from surface runoff has been calculated using the export coefficients of Rast and Lee (1978) (0.5 kg ha^{-1} yr^{-1} for rural/agricultural land, and 0.1 kg ha^{-1} yr^{-1} for forested land). The theoretical loading estimate from surface runoff calculated in this manner is presented in Tables 47 and 48. The theoretical loading estimate from surface runoff, including urban (cottage) lands is therefore estimated at 4226.6 + 106.0 = 4332.6 kg yr^{-1}. Using the same coefficients we can estimate pre-settlement loading at 2308.2 kg yr^{-1} (100% forest cover). Thus, the annual phosphorus loading may have been increased by 87.7% as a consequence of clearing 21.4% of the watershed.

Although there was no significant difference between agricultural and forest total nitrogen export coefficients, there was a statistically significant difference between the flow-weighted means for this and all other nutrients. As mentioned previously, this may be due to the apparent differences in water yields from the two land use types.

Omernik (1977) indicates that in studies of control plots where slope, soil types, and climatic conditions are similar, the areal water yield would probably be somewhat greater from agricultural plots than from forested ones because of less plant biomass in the agricultural areas, and therefore, less water loss through evapotranspiration. Unlike control plots, however, stream basins in the Baptiste watershed may be characterized by differences

Table 47. Theoretical total phosphorus and total nitrogen loadings to Baptiste Lake through streams (kg yr^{-1})

Stream Basin	Forested TP(kg)	Forested TN(kg)	Agricultural TP(kg)	Agricultural TN(kg)	Total TP(kg)	Total TN(kg)
A	12.46	124.60	362.00	1448.00	374.46	1572.60
B	0.43	4.30	28.75	115.00	29.18	119.30
C	10.56	105.60	81.35	325.40	91.91	431.00
D	48.71	487.10	86.95	347.80	135.66	834.90
E	719.60	7196.00	641.85	2567.40	1361.45	9763.40
F	321.47	3214.70	27.45	109.80	348.92	3324.50
H	4.66	46.60	10.10	40.40	14.76	87.00
I	24.47	244.70	56.35	225.40	80.82	470.10
J	8.00	80.00	15.70	62.80	23.70	142.80
K	22.93	229.30	8.90	35.60	31.83	264.90
L	553.72	5537.20	546.00	2184.00	1099.72	7721.20
M	0.06	0.60	30.90	123.60	30.96	124.20
Total	1727.07	17270.70	1896.30	7585.20	3623.37	24855.90

Table 48. Theoretical total phosphorus and total nitrogen loadings to Baptiste Lake from diffuse runoff areas (kg yr^{-1})

Diffuse	Forested		Agricultural		Total	
Runoff Area	TP(kg)	TN(kg)	TP(kg)	TN(kg)	TP(kg)	TN(kg)
1	0.16	1.60	8.55	34.20	8.71	35.80
2	0.54	5.40	25.15	100.60	25.69	106.00
3	1.25	12.50	2.25	9.00	3.50	21.50
4	0.85	8.50	-	-	0.85	8.50
5	6.27	62.70	19.10	76.40	25.37	139.10
6	0.78	7.80	1.10	4.40	1.88	12.20
7	-	-	5.55	22.20	5.55	22.2
8	0.85	8.50	19.45	77.80	20.30	86.30
9	16.04	160.40	26.95	107.80	42.99	268.20
10	2.38	23.80	-	-	2.38	23.80
11	0.39	3.90	-	-	0.39	3.90
12	-	-	345.30	1381.20	345.30	1381.20
13	58.69	586.90	61.70	246.80	120.39	833.7
Total	88.20	882.00	515.10	2060.40	603.30	2942.40

in topography, soils and climate. It is probable that the higher areal water yields from forested areas may be due to greater slopes in the western portion of the watershed. Furthermore, because of their large size, flow patterns are more stable in the forested basins in comparison to any of the agricultural ones. This stability may have allowed for a more precise estimate to be made of the annual (and therefore areal) water yield through forested basins.

The soluble reactive phosphorus/total phosphorus (SRP/TP) and total inorganic nitrogen/total nitrogen (TIN/TN) ratio analyses results indicate that the potential exists for a greater loss of biologically available nutrients from agricultural as opposed to forested stream basins, with the available portion of the total approximately doubling as the land use changes.

Omernik (1977) found in an analyses of export from 928 non-point source type watersheds in the U.S. that the inorganic nitrogen component increased from about 18 percent in streams draining forested areas to almost 80 percent in streams draining agricultural watersheds. The inorganic (SRP) portion of the total phosphorus fraction, however, remained at the 40 to 50 percent level regardless of land use type. In contrast to our findings, Minns and Johnson (1979) report that for SRP the range for all their river studies (those draining into the Bay of Quinte) has been 10 to 30 percent since 1970.

Regardless of how the total phosphorus or total nitrogen loads fluctuate in composition as a consequence of land surface disturbances, the proportionality of inorganic N: inorganic P remains below 7.2, with few exceptions. This is an important characteristic of runoff in this watershed and may be a significant determinant of phytoplankton succession as phosphorus loadings increase.

The soluble reactive phosphorus test, however, is not always a meaningful estimate of this orthophosphate concentration. Peters (1977) showed that SRP consistently overestimated orthophosphate concentrations in streams entering Lake Memphremagog, and that orthophosphate showed no consistent relation to SRP; these conclusions were also reached by Rigler (1968). Similar results were reported by Lange and Cooke (1978) who reported that the ratio of available phosphorus to SRP averaged 0.40 for a forested watershed and 0.20 for the urban.

Schaffner and Oglesby (1978) reported that when highly agricultural watersheds were compared with those dominated by forests, the use of BAP rather than total phosphorus in calculating phosphorus runoff somewhat reduced the importance of agriculture as a source. As an example, the total phosphorus in runoff from one of their largest agricultural watersheds (Conesus) was 3.2 times that from their largest forested watershed (Honeoye). Biologically available phosphorus, however, was only 2.1 times greater than that in the forest runoff. Consequently, a more detailed investigation of the biologically available fraction of total phosphorus measured in the streams of this region is required before the limnological impact of land use changes may be totally assessed.

Based on our results we may conclude that undisturbed forested stream basins in the Baptiste Lake watershed are relatively stable with regard to the loss of nutrients and that the quality of stream water is generally high. Whatever biogeochemical controls are involved in maintaining these characteristics may be altered by the clearing of forested lands.

8.9 SAMPLING CONSIDERATIONS FOR AGRICULTURAL AND FORESTED STREAMS

The frequency of sampling required for reliable estimates of nutrient loadings must be established before one can accurately evaluate the relative significance of runoff types entering a lake.

The data for both agricultural and forested watersheds at Baptiste Lake suggest that selective sampling for TN, TON, TP and SRP based upon changes in discharge during spring runoff and precipitation events would have improved estimates of annual loadings of these nutrients. If either concentration or discharge were to be measured on a random or fixed schedule, peaks might be missed, and the resulting loadings would be underestimated. Similar results were reported by Sharpley, Syers, and O'Connor (1976) who found that deviations in estimates of phosphorus loadings generally increased with an increase in sampling interval during specific runoff events in a stream draining an agricultural watershed.

Although the sampling frequency for stream "E" compares favourably with changes in discharge for that stream (Figure 66), the hysteresis effect has not been rigorously accounted for.
It is difficult to interpret many of the variations in ammonia and nitrate-nitrite concentration in agricultural and forested runoff based on the hydrological regimes alone. Levels of these nutrients are highly dependent on seasonal biological activity which in turn,

are dependent on climatic and physical conditions. In order to improve loading estimates, ammonia should be sampled more intensively during spring runoff, and during or shortly after periods of high organic nitrogen input in summer and autumn. Nitrate - nitrite should also be monitored intensively during spring runoff, during periods following peaks in ammonia, and during late autumn when concentrations are again high.

Because phosphorus and nitrogen are frequently associated with sediment both directly and also indirectly through sorption - desorption reactions (Meyer 1979; Peters 1977) their losses may be closely associated with the loss of sediment through the stream. Several studies have indicated that a large proportion of the total annual transport of sediment-related phosphorus and nitrogen can occur in relatively short periods of stormflow (Bargh 1978; Ongley et al. 1977; Lange and Cooke 1978). In order to improve our capability to predict "total" loadings, further investigations should be directed towards determining the extent to which nutrient concentrations (both in dissolved and particulate form) vary with changes in discharge for both land use types.

In large basins, discharge in response to spring snowmelt or precipitation events is more prolonged and therefore the timing of sampling may not be as critical as in streams with small basins where discharge peaks and declines very rapidly. Furthermore, because nutrient export depends strongly on the hydrological regime, more emphasis should be placed on monitoring nutrient loadings through basins which are characterized by the highest annual hydraulic discharges.

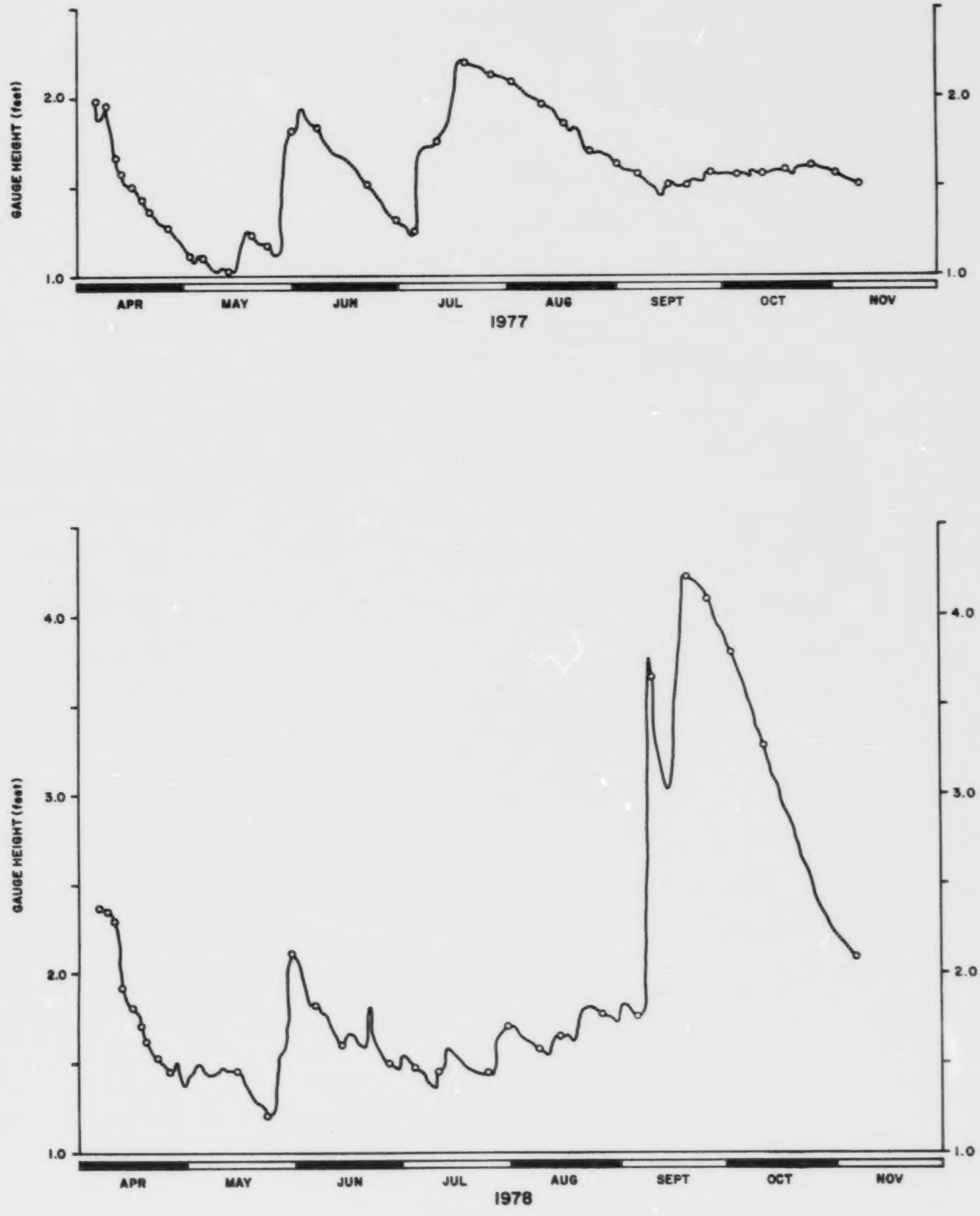

FIG. 66 SAMPLING SCHEDULE RELATED TO STAGE HEIGHT FOR STREAM "E"

8.10 TOTAL PHOSPHORUS IN THE OUTLET STREAM

According to assumptions in Vollenweider's (1969) steady state model, the total phosphorus concentration in the outflow of a lake should be equivalent to the in-lake concentration.

In general, we have observed that the pattern in the outlet is similar to that observed at the surface (0.5 m) of the north basin sampling site. In 1977, total phosphorus concentrations peak during spring and are then followed by a drop until mid July. At this time concentrations are relatively constant around 0.060 mg/L in both the lake and outflow. They began to increase after July and remain close to or slightly above 0.100 mg/L until late fall at which time phosphorus drops back to lower levels.

8.11 ATMOSPHERIC DEPOSITION

Nutrient input to Baptiste Lake via atmospheric deposition was measured in the 1978-1979 water year. Atmospheric deposition includes materials collected during a rain or snow event and the particulate material (soils, pollen) deposited between precipitation events (Scheider, Snyder and Clark 1979).

Nutrient input via precipitation and dustfall can be expected to vary seasonally and from one locale to another depending on such variables as wind, precipitation, vegetation, land use, etc. In areas of high soil disturbance, i.e. agricultural zones, dry fallout or dustfall may contribute to significantly higher inputs of nutrients per unit surface area of water than atmospheric deposition from a forest zone where land disturbance is minimal.

Rainfall and snowfall water equivalents collected in the Baptiste Lake bulk samplers were similar on most occasions at all sites, although measured concentrations and thus loadings displayed spatial and temporal variations. We conclude that particulate dustfall rates were in fact site specific and also seasonally variable. The volume-weighted mean (total mass divided by total volume) and range of TP and TN concentrations for winter and summer bulk precipitation at four sampling sites are present in Table 49. The atmospheric nutrient loadings measured during specific time intervals as well as annual areal deposition rates are presented in Table 50.

The greatest proportion of TP and TN loadings occurs during the summer months (May to August) following the annual precipitation pattern; loadings during spring, fall and winter are small in comparison. Total phosphorus deposition onto the lake via rainfall and dustfall as measured from April 20, 1978, to November 7, 1978 was 253.3 kg. Total nitrogen deposition for the same period was 3603.4 kg. Total phosphorus deposition onto Baptiste Lake via snow and dustfall as measured between November 7, 1978 and April 26, 1979 was estimated at 43.8 kg. Total nitrogen depostion during this interval was estimated at 886.1 kg. Combining these figures gives totals of 279.0 kg for total phosphorus and 4489.5 kg for total nitrogen for the 1978 - 1979 water year. Expressed in terms of deposition per unit area, this represents an annual rate of 30.4 mg m^{-2} for total phosphorus and 489.6 mg m^{-2} for total nitrogen.

During winter, nutrients in snow which accumulates on the lake ice surface are not immediately made available to the lake, and

Table 49. Chemical Composition of Atmospheric Deposition (mg L^{-1})

		Rain and Dustfall Apr. 20/78 - Nov. 7.78		Snow and Dustfall Nov. 7/78 - Apr. 26/79	
Sampling Site		x	Range	x	Range
TP	1	0.05	0.02*-0.18	0.06	0.06
	2	0.06	0.02*-0.15	0.04	0.03-0.05
	3	0.06	0.02*-0.36	0.04	0.03-0.05
	4	0.05	0.02*-0.23	-	-
TN	1	0.96	0.40-2.15	1.13	1.10-1.16
	2	1.04	0.17-2.21	0.95	0.93-0.97
	3	1.29	0.22-6.22	0.76	0.74-0.78
	4	0.98	0.24-2.04	-	-

* below detectable limit

Table 50. Nutrient inputs to Baptiste Lake via atmospheric deposition during the 1978 - 1979 water year (kg)

Interval	TN	TP
Apr 20 - May 15	425.74	18.00
May 15 - Jun 06	690.22	57.90
Jun 06 - Jul 04	519.40	58.20
Jul 04 - Jul 25	357.30	23.30
Jul 25 - Aug 14	1063.44	29.30
Aug 14 - Sep 05	285.39	39.36
Sep 05 - Sep 19	175.07	3.65
Sep 19 - Oct 11	81.94	4.70
Oct 11 - Nov 07	4.92	0.84
Nov 07 - Apr 26	886.06	43.77
Total	4489.48	279.02
Annual Deposition mg m^{-2} yr^{-1}	489.57	30.42

are deposited over a very brief time interval at ice-out. However, the actual loadings from this source at ice-out were neglible in comparison to inputs from surface runoff during the months of March and April; the latter ranged from 738 to 1232 kg for total phosphorus and from 6468 to 7321 kg for total nitrogen over the three year study period. Precipitation during the open water season contributes nutrients to the lake immediately, and it has been reported that increases in primary production are possible subsequent to summer rainfall events. For example, Brakke (1977) reported that phytoplankton growth in nitrogen-limited surface waters of a hypertrophic lake was stimulated by rainfall. We have no specific evidence to support this observation, and in any event these atmospheric inputs were still far less than loadings from surface runoff which ranged from 773 to 3756 kg for total phosphorus, and from 5583 kg to 56,062 for total nitrogen for the May to November period.

In general, nutrient loadings through atmospheric sources become more important as the watershed area decreases relative to the surface area of the lake. For example, Dillon (1974) estimated that dryfall and direct precipitation supplied between 0.3 and 79 percent of the annual phosphorus budget of eleven lakes in southern Ontario; the higher figures representing lakes with smaller drainage basins. In addition, atmospheric contributions to the total loadings are significant where chemical weathering in the lake's watershed is low, this is particularly characteristic of watersheds in the Canadian Shield where weathering rates are low due to resistant nature of the bedrock. The large size of the

Baptiste Lake watershed and the biogeochemical characteristics of the region reduces the significance of atmospheric nutrient loadings to this system.

The rates of atmospheric deposition of total phosphorus and nitrogen measured at Baptiste Lake are compared to those developed by others in Tables 51 and 52.

The only other data available for deposition of nutrients in Alberta are these of Caiazza et al. (1978). They present a total annual phosphorus deposition of 113 mg m^{-2} and a total annual nitrogen deposition of 1,617 mg m^{-2} for the Cooking Lake region southeast of the city of Edmonton. Since the sites are downwind from zones of higher industrial and agricultural activity, one might anticipate that the deposition rate would be substantially higher to those observed in the Baptiste Lake watershed. In general, our measured deposition rates are lower than those observed elsewhere. Year to year variations can be substantial and obviously the best estimates can only be derived from long-term observations (Schindler et al. 1976; Likens et al. 1977).

In order to determine which technique gave the most representative sample, Galloway and Likens (1976) investigated the different methods and designs of samplers which are presently available to measure precipitation. The Hubbard Brook type bulk rain and snow collectors (Likens et al. 1977) used in this study were rated as being reliable in terms of collection efficiency. According to the same authors, chemical and biological changes can occur in a rain sample if preservatives are not used, or if the pH is not less than 4.5. Consequently, they recommend

Table 51. Atmospheric deposition of total phosphorus.

Author	Type	Location	Rate ($mg\ m^{-2}\ yr^{-1}$)
Rast and Lee (1978)	Bulk	Review	100
Caiazza et al (1978)	Bulk	Near Edmonton	113
Schindler et al. (1976)	Bulk	ELA, Ontario	24-53
Scheider et al. (1979)	Bulk	Haliburton, Ont.	41.5 (1977)
Jansson (1979)	Bulk	Kuokkel, Sweden	5.5
Nicholls and Cox (1978)	Bulk	Harp Lake, Ont.	74.4
Dillon (1974)		Haliburton, Ont.	77.
Dillon (unpubl)	Bulk		42.
Gomolka (1975)	Bulk	Lake St. Nora, Ont.	37
Cross (1977)	Bulk	Bob Lake, Ont.	44.3
Uttormark (1974)	Bulk	Review	7-800
This study	Bulk	Baptiste Lake, Alta.	30.4

Table 52. Atmospheric deposition of total nitrogen.

Author	Type	Location	Rate ($mg\ m^{-2}\ yr^{-1}$)
Rast and Lee (1978)	Bulk	Review	2400
Caiazza et al. (1978)	Bulk	Near Edm., Alta.	1617
Schindler et al.(1976)	Bulk	ELA, Ont.	477-732
Scheider et al. (1979)	Bulk	Haliburton, Ont.	1015
Janson (1979)	Bulk	Kuokkel, Sweden	115
Nicholls and Cox (1978)	Bulk	Harp Lake, Ont.	1600
Uttormark et al. (1974)	Bulk	Review	650-3010
This study	Bulk	Baptiste Lake, Ata.	489.6

sampling on an event basis and analyzing immediately if a high degree of accuracy is required. In the Baptiste study, a three to four week period was required to collect a volume of rainfall at each site that was sufficient for the nutrient analysis. Chemical preservations were not added to the collectors prior to each sampling interval to avoid interfering with the measurement of parameters not requiring preservation. Consequently, inorganic nitrogen levels may have changed considerably through volatilization over the collection period.

Proper placement of precipitation collectors is also necessary in order to describe accurately the input of nutrients from this source. Several authors have demonstrated that annual loading figures of phosphorus are overestimated when precipitation samples are collected on land as opposed to directly over the lake surface (Gomolka 1975; Dillon 1974; Nicholls et al. 1978). This is due primarily to the contribution made by dustfall from the land area surrounding the collector. Collectors should be placed over the lake surface or as close as possible to the lakeshore if measurement of nutrient inputs directly to the lake are desired. In addition, the use of a snow collector on the lake ice surface rather than sampling the actual snow cover would avoid the problem of contamination by winter overflow.

8.12 SHORELINE DEVELOPMENT

In 1977, approximately 87% of the lakeshore dwellings at Baptiste lake were served by pit toilets, 10% by septic tanks, and 3% used chemical toilets. Wastewater disposal followed a similar pattern with approximately 84% of dwellings discharging into dry wells, pit toilets, or onto the ground surface, and 10% using septic tanks (Alberta Municipal Affairs 1977).

Using average data for the town of Athabasca in 1976 and 1977 the per capita output was calculated to be 0.93 kg capita^{-1} yr^{-1} total P and 4.40 kg capita^{-1} yr^{-1} total N. This figure for phosphorus is similar to the figure of 0.8 kg capita^{-1} yr^{-1} used by Dillon and Rigler (1975), and falls within the range of data reported in other studies (Uttormark et al 1974; Alexander and Stevens 1976). One point to note is that the figure used by Dillon and Rigler (1975) was an estimate for the physiological output of phosphorus, whereas the figure for Baptiste Lake includes all factors contributing to municipal sewage and may be more representative of the nutrient sources on the lakeshore.

"T" for cottages was estimated to be 0.66 capita - yr cottage^{-1} yr^{-1}, similar to the figure of 0.69 used by Dillon and Rigler (1975) for lakes in southern Ontario. A separate "T" value for permanent dwellings was estimated at 1.3 capita-yr permanent dwelling^{-1} yr^{-1}.

Using an area of 102.3 hectares for the lakeshore lots developed by 1977 and an export coefficient of 1.0 kg ha^{-1} yr^{-1} total P and 5.0 kg ha^{-1} yr^{-1} total N (Rast and Lee 1978) gives the following estimates for nutrient runoff:

$$\begin{aligned}\text{Total P runoff} &= 102.3 \text{ ha} \times 1 \text{ kg ha}^{-1} \text{ yr}^{-1} \\ &= 102.3 \text{ kg yr}^{-1} \\ \text{Total N runoff} &= 102.3 \text{ ha} \times 5 \text{ kg ha}^{-1} \text{ yr}^{-1} \\ &= 511.5 \text{ kg} \times \text{yr}^{-1}\end{aligned}$$

This approach furnished a maximum estimated phosphorus input of 300.0 kg yr^{-1} (1976) and 314.0 kg yr^{-1} (1977) from lakeshore dwellings. The nitrogen inputs for the same years were estimated at 1449 yr^{-1} and 1515 kg yr^{-1} respectively.

8.13 LEAF LITTER NUTRIENT LOADINGS

The direct input of leaves may be an important source of nutrients to certain lakes in the mixed boreal forest biome of Alberta, depending upon the relative magnitude of other sources. Our attempts to measure this input to Baptiste lake were minimally successful due to the destruction of most of the collectors by wind and wave action. Data are only available from one site in the north basin.

Approximately 83% of the lake's shoreline is covered by trees and shrubs; the principal species are trembling aspen (Populus tremuloides), balsam poplar (Populus balsamifera), willow (Salix sp.), birch (Betula sp.) and black spruce (Picea mariana). We believe that the majority of leaf litter on the eastern shoreline of the lake may be blown inland because of the prevailing winds from the north west, and almost no leaf litter was observed either on the lake surface or submerged along this shoreline during the collection period. We have estimated that only 56% of the shoreline could effectively contribute leaves to the lake based upon the exposure to prevailing winds. The bulk of litter that is blown onto the lake appears to fall within 9 meters of shoreline. In other lake studies it has been reported that almost no litterfall is observed beyond 10 meters offshore and that the decrease in offshore transport of this material may vary from linear to exponential depending on local conditions (Szczepanski 1965; Mathews and Kowalczewski 1969; Gaslith and Hasler 1976).

Nutrient loadings were estimated by two methods. First, by extrapolating the loadings results from the one collection site to the total length of wooded shoreline, and secondly, to that portion of shoreline which could potentially contribute litter fall to the lake as a result of prevailing wind direction.

We have estimated an annual transport of 710 gm (dry weight) of leaves per meter of wooded shoreline (79 gm m^{-2} of littoral zone). This compares favourably to the 500 gm m^{-1} reported by Szczepanski (1965), 770 gm m^{-1} reported by Gasith and Hasler (1976), and 200 - 500 gm m^{-2} reported by Levanidov in Kaushik and Hynes (1971). On a dry weight basis, leaf litter consisting of 75.3% poplar, 17.5% willow, 2.5% spruce and the remaining 3.7% consisted of unknown leaves, seeds, etc. The nitrogen and phosphorus contents of the collected leaf litter are presented in Table 53.

The estimated nutrient loadings calculated by this technique are presented in Table 54. All loadings are relatively small in comparison to the total from all other sources. Even if we assume a 20% additional transport of miscellaneous airborne litterfall through the rest of the year (Pieczynska 1972; Fisher and Likens 1973), the annual loadings would still be lower than those contributed by atmospheric deposition in winter.

Since our data are inadequate to describe the spatial variability that may exist in the transport of leaves offshore in

Table 53. Range and mean concentrations of nutrients in leaf litter (mg gm^{-1} dry weight)

	Poplar	Willow	Spruce	Misc.
TN	14.32 (10.87-18.33)	6.55 (4.41-10.60)	21.25 (16.25-23.96)	7.43 (6.53-7.91)
TP	2.80 (2.35-3.14)	1.46 (0.87-2.14))	1.19 (1.04-1.34)	2.21 (1.81-2.62)

Table 54. Estimated Leaf Litter Loadings (kg yr^{-1}) 1978

	Total Loading	Potential Loading via Prevailing Winds	
Total Nitrogen		234.05	131.36
Total Phosphorus	37.68	21.15	

different locations, further study would be useful in order to describe more accurately nutrient loadings through leaf litter input to our lakes.

Leaves must decompose before nutrients are released completely, whereas from most other sources, nutrients dissolved in the transport medium (streamwater, groundwater, rainfall, etc.) may be more immediately available for uptake by primary producers in the lake.

The abrasion of leaves by wave action, "shredding" by macroinvertebrates, and microbial decomposition are major factors which may increase the rate of leaf decomposition in the littoral zone. However, when leaves are carried further out into the lake and sink, they may become partly or completely buried in the sediment exposing little leaf surface area. Depending on the depth of the water, abrasive forces may be eliminated entirely from the decomposition process. Invertebrate and microbial effects would also be reduced if the oxygen content in the sediment-water interface was too low to support these populations (Gasith and Lawacz 1976; Cummins 1974). As a result, decomposition may proceed very slowly. Leaves were recovered frequently on anchors from depths of over 20 meters in Baptiste Lake. They were relatively intact, and the decomposition rate of many leaves entering the lake may be quite slow. Consequently, the actual contribution of nutrients to the lake that are readily available for uptake by phytoplankton may be even less than that indicated by our data.

8.14 TOTAL PHOSPHORUS LOADINGS TO GHOST AND ISLAND LAKES

The export coefficients used to determine the phosphorus contribution from surface runoff were those of Rast and Lee (1978). Atmospheric deposition phosphorus inputs were calculated using rates derived from observations at the Baptiste Lake. In order to derive a loading figure from shoreline development, the modified version of the analysis by Dillon and Rigler (1975) used on Baptiste Lake was applied. Through a questionnarie distributed to Island Lake residents the parameter "Tc" was determined to be 0.52 capita - yrs cottage^{-1} and "Tp" 1.04 capita - yrs permanent dwellings^{-1} yr^{-1}. The parameter "Nc" is equal to 140 cottages and "N_p" is equal to 16 permanent lakeshore dwellings.

Upstream lakes can act as a sink or trap with respect to watershed nutrient flux. The retention (Rp) or portion of the phosphorus input that is trapped in Ghost Lake was estimated by an empirically derived relationship with qs, the areal water load (Ostrofsky 1978).

i.e. $Rp = .201e^{-.0425qs} + .574e^{-.00949qs}$

The phosphorus loading to Ghost Lake via surface runoff was estimated to be 304.2 kg yr^{-1} and that from atmospheric deposition to be 60.5 kg yr^{-1}. The total existing load to Ghost Lake was estimated to be 364.7 kg yr^{-1} with an areal loading to the lake equal to 183.2 mg P yr^{-1} m^{-2}. The areal water load for Ghost Lake is .497 m yr^{-1} and thus Rp = 0.76.

The portion of the average total phosphorus input to Ghost Lake which is transported into the outlet creek is therefore 364.7 (1-Rp) = 87.5 kg yr^{-1}. The major phosphorus inputs to Island Lake are summarized in Table 55.

Table 55. Theoretical phosphorus inputs to Island Lake

Source	Annual Loading (Kg)
Ghost Lake	87.5
Shoreline Development	160.1
Atmospheric Deposition	236.4
Surface Runoff	1061.0
Total	1545.0
Areal Load	197.4 mg m^{-2} yr^{-1}

The theoretical phosphorus input of 595 mg m^{-2} yr^{-1} to Baptiste Lake is more than double that for either Ghost or Island Lake. This merely reflects differences in the watershed to lake surface area ratios on which the relative significance of surface runoff nutrient loading is highly dependent.

As in the Baptiste Lake study, we did not attempt to quantify inputs from other sources such as groundwater, sediment release, waterfowl, etc. Because the major portion of Island Lake is relatively shallow and probably partly stratified, inputs from sediment release may be highly significant.

9.0 PALEOLIMNOLOGY

Sediment cores were taken from the south basin in mid-March, 1977, while ice still covered the lake. A two inch diameter, modified Livingstone piston corer (Wright et al. 1965) was used to obtain the cores from a depth of 22 m. The following section is extracted from the report of that analysis (Hickman et al. 1978).

All the measured parameters indicate that Baptiste Lake and its catchment area have undergone little change during the last 4600 ^{14}C years. The extremely rapid sedimentation rate (0.18 cm yr^{-1}) has also permitted an examination of the lake's history during the period of settlement by modern man.

The pollen frequency fluctuations from Baptiste Lake are not indicative of either long-term succession, species migration or climatic change in the watershed. Combined totals of the non-arboreal pollen percentages are never high enough to suggest grassland or prairie development. All indications are that the vegetation surrounding Baptiste Lake has remained forest.

Settlement and agriculture have no doubt produced the greatest effects on the landscape. An estimated deposition rate of 0.18 cm $year^{-1}$ for Baptiste Lake places the turn of the century near 14 cm core depth. Since compaction is not nearly as great at the top of the core it would seem reasonable to place 1900 A.D. somewhat lower than 14 cm. The most noticeable change in the upper portion of the core is the drop in _Picea_ percentages from 34% at 25 cm depth to only 12% in the very top sample.

Pinus also drops in frequency while _Betula_, _Populus_, _Alnus_, and Salix all show percentage increases. The drop in Picea and Pinus frequencies may reflect selective timber cutting during historic times, while the rise of deciduous elements is in response to new successional opportunities, or even fire suppression. Land clearing and agriculture have not had a significant effect on the percentages of non-arboreal pollen types.

The large number of discrete clay bands during 3400 to 4600 ^{14}C years B.P. period indicates instability within the catchment area. Production levels, as inferred from fossil pigment levels, remained relatively constant during this period. The catchment area instability could have prevented the continued maturation of the lake by interfering with light penetration, thereby reducing the photic zone as happens in glacier-fed lakes today, and by interferring with the release of nutrients from the sediments (Livingstone and Boykin 1962). Afterwards, the catchment area became more stable and productivity, organic matter, and nitrogen gradually increased.

Production increased rapidly approximately 500 ^{14}C years B.P. and caused a subsequent enrichment of both phosphorus and nitrogen in the sediments. This increased production predates white settlement by 350 to 400 years. A marked decline in production, which occurred 300^{14}C years B.P., corresponds to an increased occurrence of clay bands. These in turn reflect an increased instability of the catchment area as discussed above.

The final increase in production occurred approximately 140^{14}C years B.P. Both phosphorus and sulphate levels declined simultaneously. This together with a rise in the iron: manganese ratio suggests decreasing oxygen concentrations in the hypolimnion (MacKereth 1966). Phosphorus is more readily released from sediments under anaerobic conditions (Mortimer 1941, 1942; Livingstone and Boykin 1962; Adams and Duthie 1976) while sulphate can be reduced to sulphide if oxygen concentrations are sufficiently low (Adams and Duthie 1976).

The diatoms found in the core indicate a eutrophic lake. Planktonic taxa dominate the entire period represented by the core, indicating that lake levels have not fluctuated markedly. The most dominant species numerically, Stephanodiscus hantzschii, is characteristic of eutrophic waters (Hustedt 1949; Bradbury 1975; Hickman 1978b). Interestingly, it is able to grow in lakes where blue-green algae develop populations of bloom proportions. It prefers, or perhaps tolerates, low light levels (Megard 1969), and this characteristic combined with its small size (10 μm diameter), and easily suspended frustule would make the species an effective competitor in water made turbid with large blue-green populations at any season. Stephanodiscus astraea is similarly characteristic of eutrophic waters (Hustedt 1949), but not those which are extremely eutrophic. Therefore, its decrease to 280^{14}C years B.P. would suggest the lake was becoming more eutrophic.

The fluctuations of Asterionella formosa, Cyclotella comta and Melosira granulata also indicate that Baptiste Lake underwent a subtle change during the last 1500^{14}C years. Between 2856 and 1460^{14}C years B.P., Asterionella formosa was an important component of the plankton. In North America, this species is most commonly found in mesotrophic to eutrophic lakes (Bradbury 1975).

Therefore, its demise could be interpreted as indicating a change toward increased eutrophy. Such conditions would then have favoured the development of Cyclotella comta. As the lake continued becoming more eutrophic, this species also declined to be replaced by Melosira granulata, which grows well in strongly eutrophic waters and exists in a pH range of 7.8 - 8.2 or greater (Schoemann 1973; Hickman 1974). Therefore, during the last 1500^{14}C years Baptiste Lake has steadily become more eutrophic. This trend appears to have accelerated approximately 250 years ago.

Another factor indicating the productive nature of Baptiste Lake is the dominance of but a few diatoms; the more productive lakes generally possess fewer species than less productive ones and this is reflected in the sedimentary record (Bradbury and Waddington 1973).

In summary, the data obtained from the Baptiste Lake sediments indicate that the vegetation in the catchment itself has changed over a 4600 ^{14}C year period. The lake has always been eutrophic and approximately 1500 ^{14}C years ago it became even more eutrophic as shown by changes in the dominant diatoms; this trend accelerated approximately 250 years ago. The lake, therefore, is eutrophic as are many others in central Alberta.

10.0 THE PHOSPHORUS MODELLING SEQUENCE APPLIED TO BAPTISTE LAKE

One of the principal objectives of this study was to examine the methodologies available for quantitatively predicting the impact of shoreline and watershed development on lake trophic status. These methods have generally been based upon phosphorus concentration-phytoplankton standing crop relationships, the assumption being that phosphorus is the principal factor limiting the volumetric concentration of phytoplankton biomass that will develop in most lakes. The following discussion will briefly review the quantitative methods presently available for describing phosphorus dynamics in lake systems. When appropriate, these methods will be tested against the observed Baptiste Lake data in order to gain insights into factors controlling phosphorus dynamics in the lake and its relationship to primary production.

The phosphorus modelling sequence consists of three basic components:

a) the prediction of algal standing crop from a knowledge of in-lake phosphorus concentration.

b) the determination of mean in-lake phosphorus concentration either through observation, or by modelling with a knowledge of phosphorus loading, hydrology and morphometry of the water body in question, and

c) the determination of phosphorus loading to the water body either by observation, by modelling with aid of known nutrient export coefficients and atmospheric deposition coefficients, or as is illustrated below by back calculating using the simple mass balance and steady state loading models.

10.1 METHODS FOR PREDICTING PHYTOPLANKTON STANDING CROP

Nicholls and Dillon (1978) have recently reviewed several phosphorus-chlorophyll relationships published in the literature. Upon close examination one observes that a wide range of chlorophyll a concentrations can be predicted from a single phosphorus concentration, depending upon which predictive model is used. The authors discuss the major sources of this variation, and these include (a) the relative bio-availability of different phosphorus fractions in different lakes (total phosphorus is used in all but one relationship), (b) the inherent variability of cellular chlorophyll a contents, and (c) differences in sample collection and analysis procedures. As noted in their paper, three of these models include maximum values for one or both of the parameters, four relationships essentially use spring overturn phosphorus with mean summer chlorophyll, and nine others use some combination of summer or annual mean values.

We have tested these relationships with the 1977 Baptiste Lake data in order to determine their utility in predicting phytoplankton standing crops in a eutrophic situation (Table 56) and have found that three of the spring phosphorus-summer chlorophyll models, i.e. Dillon and Rigler (1974), Edmonson (1972), and Oglesby, Schaffner, and Mills (1975) produced an excellent agreement between predicted and observed values; one should note, that large confidence limits could be attached to these correlations. The fourth one, that of Sakamoto (1966), gave a much higher figure than was actually observed, probably due to the inclusion of a portion of chlorophyll b and c in his original pigment analysis

Table 56. Comparison of predicted vs. observed chlorophyll (mg m^{-3}).

Reference	Predicted	Observed
Sakamoto (1966)	75.6	41.2
Lund (1970)	69.9	196.0
Brydges (1971)	15.4	34.5
McColl (1972)	19.1	55.4
Edmonson (1972)	43.8	46.8
Megard (1972)	44.8	55.4
Dillon & Rigler (1974)	41.6	41.2
Nusch (1975)	20.9	196.0
Oglesby et al. (1975)	43.0	46.8
N.E.S. (1974)	25.9	22.5
Schindler (1976)	28.3	22.5
Jackson (1976)	24.5	31.7
Jones and Bachman (1976)	40.2	41.2
Nicholls (1976)	67.4	21.7
Carlson (1977)	41.0	55.4
Nicholls (1977)	18.8	34.5
Schindler (1978)	24.6	22.5

(Nicholls and Dillon 1978). The best prediction of these four was given by the Dillon and Rigler relationship, i.e.

$$\mathrm{Log}_{10}\,[B] \;=\; 1.45\,\mathrm{Log}_{10}\,[TP] - 1.14$$

Several other methods also gave good results, including those of Schindler (1976) and the National Eutrophication Survey (1974) which predict mean annual chlorophyll a concentration from mean annual and median annual TP respectively. The models of Megard (1972), Jackson (1976), and Carlson (1977) which compare seasonal (summer) phosphorus and chlorophyll a concentrations also gave reasonable results. Three other seasonal models gave poor results: Brydges (1971), McColl (1972), and Nicholls (1977). The weakest of all were those of Lund (1970), Nusch (1975), and Nicholls (1976) which incorporate a seasonal maximum of one or both parameters into the correlation. Finally, the relationship generated by Schindler (1978) in a synthesis of global data on phytoplankton-nutrient relationships:

$$\mathrm{Log}_{10}\,[B] \;=\; 1.213\,\mathrm{Log}_{10}\,[TP] - 0.848$$

provided a close prediction of 24.6 mg m^{-3} chlorophyll a compared to the observed annual mean of 22.5 mg m^{-3}.

This analysis indicated that the observed Baptiste Lake standing crop data could be quantitatively described by many of the existing empirical models, and implied a strong dependence of summer phytoplankton growth on phosphorus concentrations. Prepas and Trew

(1983) have recently confirmed the general correspondence between total phosphorus and chlorophyll a concentration in lakes situated in central Alberta including Baptiste Lake. However, because of internal phosphorus loading which occurred during summer in many lakes, summer TP and not spring TP was the best predictor of summer chlorophyll a. The authors also speculated on the attentuating influence of salinity and turbidity on the [chl a] : [TP] ratio, and concluded that separate models may have to be developed for the wide range of lake types in this province.

The close correspondence between standing crop and phosphorus in Baptiste Lake is also interesting in view of the fact that a transient period of nitrogen limitation is suggested by the dominance of potential nitrogen fixers and low "available" N:P ratios during mid to late summer (Table 19). These data suggest that the lake has been able to obtain nitrogen from other sources in order to permit phosphorus-proportional growth to occur (Schindler 1976); an analysis of N:P ratios can offer a partial insight into this phenomenon.

Annual mass ratios of total nitrogen to total phosphorus were calculated for all measured inputs (i.e. streams and precipitation) and the outlet of Baptiste Lake for the three years of study (Table 57). In all cases the ratio is higher in the outlet than in the inputs, and this would indicate either an increase in nitrogen concentration or a decrease in phosphorus concentration, or both, during their respective passages through the lake. An analysis of total N:P ratios as found in the top 100 cm of sediment core

Table 57. Total N:P Ratios.

	1976	1977	1978
Inputs	7.2	10.9	13.1
Outputs	13.1	27.3	19.5

Table 58. Total phosphorus inputs (kg yr^{-1})

	1976	%	1977	%	1978	%
Streams	1708.59	68.5	2336.12	68.6	4626.34	80.6
D.R.A.	186.44	7.5	456.66	13.4	498.35	8.7
Precipitation	279.02	11.2	279.02	8.2	279.02	4.9
Cottages	300.	12.0	314.	9.2	314.	5.5
Leaf Litter	21.15	0.8	21.15	0.6	21.15	0.4
	2495.20	100.	3406.95	100.	5738.86	100.
Areal Loadings (mg P m^{-2} yr^{-1})	272		372		626	

indicates a mean proportion of approximately 11.7 (n = 20) which is similar to that of the apparent incoming load, although the small inorganic fraction of TN could be lost be denitrification or volatilization (Murphy and Brownlee 1982). The high N:P ratios in the outlet may reflect an increased input of nitrogen from some other source which could be from nitrogen fixation, sediment exchange, groundwater, or some combination of all three.

Total N:P ratios in the surface waters of the south basin showed a steady increase throughout the summer, peaking in early August at the time when "available" N:P ratios are approaching their lowest value (Table 19). Since the basin is thermally stratified, the hypolimnetic buildup of inorganic nitrogen, principally in the form of ammonia, may only be made available very slowly to surface waters. Closer examination reveals that organic nitrogen reaches it's highest concentration at this time as does phytoplankton biomass, and this in turn may reflect some nitrogen fixing activity. The significance of groundwater to this basin of the lake remains unknown.

The surface waters of the north basin follow a similar pattern with total N:P ratios and organic nitrogen peaking in late July/early August, although the concentrations involved are higher at this location (Figures 26 & 30). These levels may be due to irregular circulation of nutrient-rich bottom waters into the euphotic zone as a result of water column instability and wind mixing; this nitrogen could arise from either sediment exchange processes or groundwater, as noted earlier.

Finally, stream N:P ratios remain relatively low during the summer in contrast to the high increases observed in the lake. Precipitation analyses indicate occasionally high N:P ratios, but the mass of nitrogen falling onto the lake surface during July and August from that source appears insufficient to alter surface concentrations significantly.

These observations suggest that nitrogen limitation, in terms of a seasonal phenomenon, may not significantly interfere with phosphorus proportional growth due to the success of N-fixing blue-green species in the lake.

10.2 METHODS FOR ESTIMATING STEADY STATE PHOSPHORUS

Steady state or mean annual phosphorus concentrations have been defined in various ways. According to the Vollenweider (1969) steady state model, as modified by Dillon and Rigler (1974), the steady state [P] concentration is equivalent to the mean outflow concentration, i.e.

$$[P] = \frac{J\,(1-R)}{Q}$$

$$= \frac{O}{Q}$$

where J = phosphorus input mass (kg)

R = retention coefficient

Q = discharge volume of outlet (m^3)

O = mass of phosphorus in outflow (kg)

$(1-R)$ is the fraction of the total input mass (J) which is not retained in the system.

An examination of the Baptiste Lake outflow data for the periods March, 1977 to November, 1977, and March, 1977 to March, 1978 (the former corresponding to the extent of lake sampling conducted that year) indicates the same mean phosphorus concentration of 0.070 mg L^{-1} for both periods. This in fact is a flow-weighted mean obtained by dividing the loading for that interval by the correponding discharge.

Steady state phosphorus can also be determined by in situ observations, and the great variety of techniques employed have been reviewed by Nicholls and Dillon (1978), as noted above. In terms of sampling times, these approaches have included late winter or spring overturn averages, annual averages, and averages during the period of summer stratification; a variety of sampling depth intervals is also indicated. Furthermore, the mean annual concentration of total phosphorus was assumed to be equivalent to the spring phosphorus concentration by Dillon and Rigler (1975), although in testing this assumption they observed a discrepancy between these two parameters at higher concentrations. Chapra and Tarapchuk (1976) have found that a factor of 0.9 was suitable to reduce spring concentrations to mean annual concentrations. Prepas and Trew (1983) have shown that neither assumption can be generally applied to Alberta lakes because of the large variations in spring:summer [TP] ratios.

Accurate spring phosphorus data are not available for Baptiste Lake since too few samples were taken to apply volume-weighting, and analysis problems at ice-out of 1977 further weaken the results. Sampling conducted at the Baptiste North station for the interval of March, 1977 to November, 1977 produced a mean concentration of 0.070 mg L^{-1} at both 0 and 6.0 m depth, similar to that observed in the outlet stream. The mean for the same period at Baptiste South was 0.080 mg L^{-1} at 0 m depth. This suggests that surface waters contribute most of the discharge (the strata < 6 m depth comprises > 50% of total lake volume) particularly during stratified periods, and that the mean outflow concentration is in fact representative of in-lake epilimnetic conditions. A figure of 0.070 mg L^{-1} has therefore been selected for the observed steady state phosphorus concentration for 1977.

An alternate approach to the measurement of steady state phosphorus is to calculate this parameter using the steady state models of Vollenweider (1969), Dillon and Rigler (1974), or Chapra (1975). This would assume that loading data are available either by observation or by calculation (as in Dillon and Rigler 1975).

A preliminary attempt at an inventory of the principal sources of phosphorus was carried out at Baptiste Lake and the final input figures for 1976 - 1978 are presented in Table 58; the sources measured include surface runoff via streamflow and diffuse runoff areas, atmospheric deposition, an estimate from shoreline development, and litterfall. Although an estimate of total water input via groundwater is available, the chemical data are inadequate to permit the estimation of a loading. The derivation of these loadings have been discussed in detail above, and represent the consecutive period from March 16 to March 15 of the following year. Potential sources not included are groundwater, sediment exchange, and waterfowl.

Surface water inputs contributed approximately 70 to 90 percent of the total measured phosphorus loading during the three study years (Table 58). The annual export of phosphorus through surface runoff is strongly dependent upon the annual discharge which, in turn, is dependent upon the size of the watershed. Consequently, the large size of the Baptiste watershed in comparison to the lake's surface area, coupled with the biogeochemical characteristics of the region, reduce the significance of the loadings from the other measured sources.

As previously discussed, the annual phosphorus export and water discharge pattern for the Baptiste Lake watershed is tied closely to the relationship of these two variables that exists for the undisturbed forested stream basins, and may be highly predictable. Frequency analysis can then be used to determine the 1:25 year high, low, and long-term average loadings to the lake. To accomplish this, estimated inflow runoff volumes to the lake for the period 1967 to 1978 (Table 11) were combined with an average total phosphorus flow-weighted mean concentration, and the resulting loadings were plotted on frequency curves. The weighted means were determined by adding all tributary and diffuse runoff loadings and dividing by the corresponding sum of measured discharges for each year, then, averaging them over the three year period. Annual means for total phosphorus (1976 = 0.16 mg L^{-1}, 1977 = 0.20 mg L^{1}; 1978 = 0.16 mg L) were relatively constant and were not significantly different from the three year average annual flow-weighted means developed for forested basins.

Based on these results, only long-term hydrological data are required from the Baptiste watershed to identify the export patterns of total phosphorus through surface runoff. The estimated 1:25 year high, low, and long term average phosphorus loadings to Baptiste Lake through surface runoff are equal to 5142, 594, and 2061 kg yr^{-1}, respectively.

A frequency curve analysis was also performed on the precipitation data. The annual total phosphorus loading to Baptiste Lake from atmospheric deposition during the period November 1977 and November 1978 was divided by the estimated precipitation volume input and the resulting volume-weighted mean was combined with estimated annual precipitation volume inputs to the lake for the period 1967 to 1978. The 1:25 year high low, and long-term average phosphorus loads to Baptiste Lake through atmospheric deposition are equal to 367, 212, and 286 kg yr^{-1} respectively.

If total phosphorus inputs from leaf litter and cottage activity were to remain constant, and if the estimated long-term average precipitation loadings were included as part of the total load, the long-term average surface runoff input would still contribute approximately 77 percent of the known total loading to the lake.

An alternate method of deriving nutrient inputs via surface runoff and atmospheric deposition is to calculate theoretical loadings by using nutrient export co-efficients obtained from the literature. The main drawback in using these theoretical values is that the quantity and quality of nutrient runoff is site-specific and influenced dramatically by such factors as vegetation, soil structure, topography, precipitation, etc.

As discussed previously, the phosphorus export co-efficients derived for the Baptiste Lake watershed are very close to those provided by Rast and Lee (1978). The theoretical value for this input to Baptiste Lake, calculated using their co-efficients of 0.1 kg ha^{-1} yr^{-1} for forested land, 0.5 kg ha^{-1} yr^{1} for rural (agricultural) land, and 1.0 kg ha^{-1} yr^{-1} for urban (cottage) land is 4329.0 kg yr^{-1}.

The atmospheric deposition of total phosphorus on to Baptiste Lake (30.4 mg m^{-2} yr^{-1}) appears to be lower than that presented in the reviews of either Rast and Lee (1978) or Uttormark et al (1974). The data at Baptiste Lake are based on only one year of sampling whereas the data furnished by Rast and Lee (1978) represented the synthesis of many years of observations by many workers. For the sake of this discussion, we have selected Rast and Lee's (1978) theoretical value of 100 mg m^{-2} yr^{-1} for total phosphorus and an estimated input of 917.0 kg yr^{-1} is derived. Finally, a figure of 314 kg yr^{-1} was incorporated to account for domestic effluents (Trew et al. 1978). A total theoretical input of 5454.0 kg yr^{-1} or 595 mg m^{-2} yr^{-1} is derived from these calculations, which is higher than that actually measured during 1976 and 1977, and less than that of 1978.

Once the loading data have been obtained, the various steady state models can be used to predict lake phosphorus concentrations. The steady-state models presently available for expressing the phosphorus dynamics of a water body have been derived from the simple mass balance equation for non-conservative substances.

For a given time period:

change in (P) mass = INPUTS - OUTPUTS VIA OUTLET - SEDIMENTATION

or

$$\Delta M = I - O - S$$

The steady state solution to this equation is given by Vollenweider (1969) as:

$$[P] = \frac{L}{z(\sigma + \rho\omega)}$$

where $[P]$ = steady state phosphorus concentration (mg m^{-3})

L = annual areal loading (mg m^{-2} yr^{-1})

z = mean depth (m)

$\rho\omega$ = flushing rate (yr^{-1})

σ = sedimentation coefficient (yr^{-1})

This model is based on four assumptions: the sedimentation rate is proportional to the amount of the substance in the lake; any substance is completely mixed throughout the entire lake; the concentration in the outflow is equal to the concentration in the lake; there are no seasonal fluctutations in loading rate. Obviously few of these simplifying assumptions will hold for stratified temperate lakes, or for highly productive systems, but they may hold for unproductive lakes in smaller watersheds (i.e. with smaller loadings).

The year 1977 was representative of long-term water inputs. Consequently the observed loading data for the year have been combined with water renewal parameters derived from the long-term hydrological data. This is to allow for the incorporation of evaporation estimates which are unavailable for the short term (see discussion below). Solving the model for 1977 loadings requires the definition of $p\omega$ and σ. The flushing rate $p\omega$ is equivalent to the total discharge (via surface water, groundwater, and evaporation) divided by lake volume.

$$p\omega = \frac{21{,}383{,}720 \text{ m}^3 \text{ yr}^{-1}}{85{,}525{,}956 \text{ m}^3} = 0.25 \text{ yr}^{-1}$$

The sedimentation coefficient σ is equivalent to the mass of phosphorus sedimented per year divided by the mass of phosphorus in the lake. According to Hickman et al. (1978) the net sedimentation rate at the deepest site in Baptiste Lake is approximately 1.8 mm yr^{-1}. This site may be affected by slumping or focussing phenomena (Lehman 1975) which would increase the apparent accumulation, but it is also in the basin which receives approximately 80% of surface water and, hence, a higher proportion of the allochthonous phosphorus load. This was the only location cored, so there are no alternative data to use.

Annual input per sq. meter $= .18 \text{ cm yr}^{-1} \times 100 \text{ cm} \times 100 \text{ cm}$
$= 1800 \text{ cm}^3 \text{ m}^{-2}$

Density of dry material $= .328 \text{ gm cm}^{-3}$

Dry weight of annual sediment $= .328 \text{ gm cm}^{-3} \times 1800 \text{ cm}^3 \text{ m}^{-2}$
$= 589.8 \text{ gm m}^{-2}$

Core phosphorus conc. (at 10 cm) $= 935.2 \text{ mg kg}^{-1}$

Mass of phosphorus in annual sediment input per sq. meter $= 935.2 \text{ mg kg}^{-1} \times .5898 \text{ kg m}^{-2}$
$= 551.5 \text{ mg m}^{-2}$

Annual phosphorus sedimentation = $(551.5 \text{ mg m}^{-2} \times 9{,}170{,}176 \text{ m}^2) \div 10^6 \text{ mg kg}^{-1}$(based on total lake surface area)

= 5057.4 kg

Total mass of phosphorus in lake = $85{,}525{,}956 \text{ m}^3 \times 0.070 \text{ gm m}^{-3} \text{ gm kg}^{-1}$

= 5986.8 kg

Assuming that the steady state phosphorus concentration observed in 1977 is representative of the long-term situation, and realizing that the sedimentation estimate based on core data is in fact a long-term estimate, the coefficient of sedimentation can be calculated as:

$$\sigma = \frac{5057.4 \text{ kg}}{5986.8 \text{ kg}}$$

$$= .85$$

The predicted steady state phosphorus concentration using the observed 1977 loading is therefore:

$$[P] = \frac{.369}{9.3(.25 + .85)}$$

$$= .04 \text{ mg L}^{-1}$$

which is lower than the concentration observed. Solving the equation using the theoretical loading gives a steady state concentration of .06 mg L^{-1}.

Dillon and Rigler (1974) attempted to facilitate the solution of this equation by introducing a factor for phosphorus retention, Rp, which they suggested could be obtained more easily than representative sedimentation rates:

$$[P] = \frac{L\,(1-Rp)}{z\ p\omega}$$

where $[P]$ = steady state phosphorus concentration (mg m^{-3})

L = areal load (mg m^{-2} yr^{-1})

Rp = retention co-efficient

z = mean depth (m)

$p\omega$ = flushing rate yr^{-1}

The retention co-efficient is best found from field measurements as:

$$\frac{\text{INPUTS - OUTPUTS}}{\text{INPUTS}}$$

Such Measurements are difficult to define when unquantified internal or diffuse sources are present, as is frequently the case. In order to derive a loading estimate using this model, one must define Rp using one of the many empirical relationships to be found in the literature (Kirchner and Dillon 1975; Larsen and Mercier 1976; Chapra 1975; Ostrofsky 1978). We have chosen that of Ostrofsky (1978) which has been defined for lakes where retention could be measured and which was expressed as a function of qs, the annual areal water load:

$$Rp = .201\ e^{-.0425qs} + .574\ e^{-.00949qs}$$

The major shortcoming of all of the several existing empirical retention models is that all have had variable success in predicting retention at low areal water loads (Widger and Kimball 1976). The model presented by Ostrofsky (1978) gives theoretically improved estimates of retention (and consequently phosphorus concentrations) at areal water loads of less than 10 m.

The areal water load for Baptiste Lake in 1977 is as follows:

$$qs = \frac{\text{surface runoff + precipitation + groundwater}}{\text{surface area}}$$

$$= \frac{21{,}383{,}720 \text{ m}^3 \text{ yr}^{-1}}{9{,}170{,}176 \text{ m}^{-2}}$$

$$= 2.33 \text{ m yr}^{-1}$$

$$\text{thus } Rp = .201\ e^{-.09903} + 574\ e^{-.02211}$$

$$= .201\ (.0957) + .574\ (.9781)$$

$$= .1820 \qquad + .5614$$

$$= .74$$

Solving Dillon and Rigler's steady state equation using the observed 1977 loading data gives the following:

$$[P] = \frac{.369(1 - .75) \text{ mg m}^{-2} \text{ yr}^{-1}}{9.3 \times .25 \quad \text{m yr}^{-1}}$$

$$= .05 \text{ mg L}^{-1}$$

The theoretical loadings calculated according to Rast and Lee (1978) produce a steady state phosphorus estimate of .08 mg/L.

Chapra (1975) has presented an alternate, semi-theoretical relationship to calculate retention:

$$Rp = \frac{\nu}{\nu + qs}$$

where qs is the areal water load and ν is the apparent settling velocity of total phosphorus, which is the total effect of all in-lake processes in removing the material from the water column. This can be combined with the Dillon Rigler (1974) steady state model as presented above to give

$$[P] = \frac{L}{\nu + qs}$$

Solving this equation for 1977 and using an apparent settling velocity of 16 m yr^{-1} (Chapra 1975) gives a steady state phosphorus concentration of:

$$[P] = \frac{.369}{16 + 2.33} \; \frac{mg\ m^{-2}\ yr^{-1}}{m\ yr^{-1}}$$

$$= .02\ mg\ L^{-1}$$

The principal assumption here is that ν is constant for all lakes, which is unrealistic. Two other values for ν can be found in the literature. 10 m yr^{-1} in Vollenweider (1975) and 12.4 m yr^{-1} by Dillon (cited in Chapra and Tarapchak (1976)) and these give predicted steady state phosphorus concentrations of .03 mg L^{-1} and .03 mg L^{-1} respectively.

The range of steady state phosphorus concentrations derived in the preceding analyses using both the Vollenweider/Dillon and Rigler models with observed and theoretical loadings are summarized in Table 59. With only one exception, all steady state models

Table 59. Observed and predicted steady state phosphorus.

	1977 Loadings	Theoretical Loadings (mg L^{-1})
Observed	.07	
Vollenweider Model	.04	.06
Dillon and Rigler Model	.05	.08
Chapra Model ($\nu = 16$)	.02	.03
($\nu = 12.4$)	.03	.04
($\nu = 10$)	.03	.05

underestimated the observed lake phosphorus concentration. This suggests systematic errors in the analyses which could arise from a number of sources:

(a) the estimated in-lake steady state phosphorus concentration (unrepresentative sampling)

(b) the models themselves (i.e. applying steady state, single reactor models to dynamic, stratified lakes)

(c) the substance removal parameters (sedimentation, retention, settling)

(d) the hydraulic/morphometric parameters

(e) the loading estimates (excludes sediment release, groundwater)

The influence of spatial heterogeneity in the particulate component of the phosphorus pool (i.e. phytoplankton cells) has been discussed previously. The extreme vertical gradient in [TP] also presents a major problem in defining [TP] content accurately.

We have acknowledged the theoretical limitations of the steady state approach in terms of its application to stratified temperate lakes. It seems clear that for a productive, complex, and dynamic ecosystem such as Baptiste Lake, a more sophisticated model is required to predict [TP] accurately.

The values assigned to the substance removal parameters will vary between lakes and in some cases at different sites within lakes and, consequently, can contribute to errors in the solution of the models. We have used an observed sedimentation coefficient in solving the Vollenweider (1969) relationship, but the location used is probably not representative of the whole lake. The retention coefficient and settling velocities have been derived from empirical observations elsewhere which may not be applicable to Baptiste Lake.

A factor for annual water renewal is also incorporated into each of the preceeding models. The solutions to the Vollenweider (1969) and Dillon and Rigler (1974) models incorporate $z \cdot p\omega$, whereas the Chapra (1975) model utilizes qs, the areal water loading. These parameters are used interchangeably and are theoretically equivalent, as shown by Dillon and Kirchner (1975):

$$qs = \frac{Qi}{A} = \frac{z \times Qo}{V} = z \cdot p\omega$$

where Q_i = inflow volume

Q_o = outflow volume

and $Q_o = Q_i$ is assumed

We have observed some variability between these parameters in our own data (Table 12) and also in the way they have been defined in the literature. Several authors defined Q as the annual discharge volume (Dillon and Kirchner 1975; Chapra and Tarapchak 1976; Scheider 1978). Others have defined Q as the annual inflow volume (Larsen and Mercier 1976; Rast and Lee 1978). The underlying assumption is that the differences between inflows and outflows are very small. This may not always be the case depending on year to year variations in precipitation and evaporation, and depending on the significance of these process in the annual water balance of the lake in question.

Dillon and Rigler (1975) have suggested the incorporation of factors for precipitation and evaporation in lakes with small watersheds, the implication being that this source of water to the lake surface is significant in some situations.

$$Q = Ad \,.\, r + Ao\,(Pr - Ev)$$

where Ad = watershed area m^2

r = annual unit runoff $m\ yr^{-1}$

Ao = lake surface area m^2

Pr = mean annual precipitation (m)

Ev = mean annual evaporation (m)

Groundwater is estimated to contribute approximately 14% to long-term water inputs to Baptiste Lake, and approximately 15% to long-term water outputs. The phosphorus content of the groundwater in the channels may have been sufficient to contribute significantly to the load. In addition, the dramatic increases in hypolimnetic [TP], and the corresponding changes in epilimnetic [TP] are compelling evidence for significant internal nutrient regeneration in Baptiste Lake. Thus, the total [TP] loading, both observed and theoretical, may be underestimated.

The following section will assess the alternative approaches to generating phosphorus loads.

10.3 METHODS FOR ESTIMATING TOTAL PHOSPHORUS LOADS

The most direct approach, although the most time consuming and expensive one, is to measure all phosphorus inputs to the lake under study. This is obviously not feasible in every situation, and even if it were, there are numerous potential sources that are technically difficult to quantify. As indicated, several sources have not been quantified for Baptiste Lake and the preceding analysis suggests that the work carried out may have underestimated the true loadings. It is difficult to verify the accuracy of loading estimates, although Rast and Lee (1978) have provided a theoretical approach which may be used.

This technique compares the mean lake phosphorus concentration : influent total phosphorus concentration ratio with an expression of the hydraulic residence time i.e.

$$\frac{[P]}{[P_i]} = \frac{1}{1 + \sqrt{\tau\omega}}$$

where [P] = steady state total phosphorus concentration (mg m^{-3})

$[P_i]$ = influent total phosphorus concentration (mg m^{-3})

$\sqrt{\tau\omega}$ = hydraulic residence time (yr)

The mean influent concentration is calculated as:

$$[P_i] = \frac{3{,}385{,}800{,}000 \text{ mg}}{21{,}184{,}187 \text{ m}^3}$$

$$= .160 \text{ mg m}^{-3}$$

Therefore $[P] = \frac{.070}{.160} = 0.44$

The hydraulic residence time for 1977 has previously been estimated

as 4.0 years, therefore the expression

$$\frac{1}{1 + \sqrt{\tau\omega}} = \frac{1}{1 + 2.00} = 0.33$$

Any deviation from a 1:1 relationship between these ratios is assumed to indicate an underestimate or an overestimate of the loadings. The discrepancy observed in fact suggests that the estimated lake phosphorus concentration is higher than can be explained on the basis of the measured inputs alone (note: groundwater has been included in the residence time expression but not as part of the mean influent phosphorus concentration). Reworking this equation so that [P]/[Pi] is equivalent to .33 would require an annual loading of approximately 4443.9 kg. This evidence suggests that those sources which have not been quantified may be significant.

Few empirical relationships are available for determining loading estimates. One approach consists of using a correlation between chlorophyll a and Vollenwieder's (1975) phosphorus loading characteristic as modified by the results of the O.E.C.D. project (Rast and Lee 1978):

$$\mathrm{Log}\,[B] = \frac{.76\ \mathrm{Log}\ L/qs}{1 + \sqrt{z/qs}} - .259 \quad (r = 0.77)$$

where L = total phosphorus loading ($\mathrm{mg\ m^{-2}\ yr^{-1}}$)

qs = hydraulic loading ($\mathrm{m\ yr^{-1}}$)

z = mean depth (m)

A comparison of predicted versus observed chlorophyll concentrations for the lake may then be used to assess the reliability of the available loading data.

When the relevant morphometric/hydraulic data are incorporated with the 1977 loading data a predicted mean chlorophyll a concentration of 11.3 mg $\mathrm{m^{-3}}$ is obtained (no time frame indicated). The observed annual epilimnetic mean for both basins of

Baptiste Lake for January to November that year was 22.5 mg m^{-3}. The discrepancy between these two values indicates that the loading estimate for that year is suspect, (i.e. a higher input of phosphorus would be required to produce the observed standing crop).

If one back-calculates the input required to produce the observed standing crop, a figure of 8404.9 kg yr^{-1} is derived. An alternate expression of the chlorophyll concentrations, (i.e. an ice-free seasonal mean concentration of 35.0 mg m^{-3} (0.5 m)) would produce an even higher estimate of the required loading.

The detailed analysis by Schindler (1978) of relationships between various expressions of phosphorus loadings and resulting phosphorus concentration, phytoplankton standing crop and phytoplankton production values for shield lakes has provided another method of back calculating loads if mean annual [TP] and outflow volume are known. The incorporation of the volume of outflow (i.e. inflow plus precipitation minus evaporation) renders the function Ip/Vo similar to the mean annual influent phosphorus concentrations of Rast and Lee (1978) with a correction for the effects of evaporation:

$$[P] = .081 \frac{Ip}{Vo} + 8.46$$

where Ip = total phosphorus input (mg)

Vo = outflow volume (m^3)

[P] = mean annual concentration of phosphorus (mg m^{-3})

We can augment the outflow volume for Baptiste Lake with an estimate of loss via groundwater (both over the long-term) and derive:

$$
\begin{aligned}
Ip &= \frac{Vo\ (P - 8.46)}{(.081)} \\
&= 15{,}831{,}720\ \frac{(70 - 8.46)}{(.081)} \\
&= 12{,}028{,}198{,}100\ \text{mg yr}^{-1} \\
&= 12{,}028.2\ \text{kg yr}^{-1}
\end{aligned}
$$

This estimated loading is substantially higher than that observed during 1977.

The simple mass balance described in the previous section can be re-arranged to provide another method for estimating total inputs per unit time.

$$\text{i.e. } I = \Delta M + O + S$$

This approach automatically includes all sources of phosphorus, including diffuse sources such as groundwater and sediment release. The principal disadvantages are that groundwater seepage out of the lake should be incorporated in the balance (a technically difficult problem) and that the determination of a representative phosphorus sedimentation rate would require the analysis of cores taken from several locations in the lake. It may also be complicated by the vertical migration of phosphorus in the sediments (Carignan and Flett 1981).

In relatively undisturbed watersheds ΔM may be very small over the medium to long-term. Thus it may be possible to approximate the long-term input by examining the long-term sedimentation rate and extrapolating a long-term output from intensive monitoring of the outlet stream. An estimate of total water output could be derived by balancing with long-term water inputs and this could be combined with average TP concentration in the outlet observed over the shorter term. Using hydrological data presented above, the long-term total water input can be estimated at 21384 dam^3 yr^{-1}. The long-term evaporation of 610 mm yr^{-1} would remove approximately 5,502 dam^3 yr^{-1}. Therefore, the net water output assuming no changes in lake level over the long-term would be 15882 dam^3 yr^{-1} (includes our groundwater estimate). Combining this with steady state total phosphorus concentration of .070 mg L^{-1} gives a long term output of 1111.7 kg yr^{-1}. Thus the long-term loading for the lake could be estimated at 5057.4 + 1111.7 = 6169.1 kg yr^{-1}.

An alternative to this approach is to use Vollenweider's (1969) "steady state" model to back-calculate the loading required to produce the observed phosphorus concentration. This phosphorus concentration could theoretically be derived from either the lake or the outflow according to the basic assumptions in the model described earlier. Therefore:

$$[P] = \frac{L}{z\ (p\omega + \sigma)}$$

$$\text{and } L = [P] \cdot z \cdot (p\omega + \sigma)$$

$$= .07 \times 9.3\ (.25 + .85)$$

$$= .716\ \text{gm m}^{-2}\ \text{yr}^{-1}$$

$$\text{or}\ \ 6566.7\ \text{kg yr}^{-1}$$

Similarly, the Dillon and Rigler (1974) model can also be re-arranged to provide an estimate of the required loadings:

$$[P] = \frac{L\ (1-R)}{z \cdot p\omega}$$

$$L = \frac{(P) \cdot z \cdot p\omega}{1 - Rp}$$

$$= \frac{.07 \times 9.3 \times 2.5}{1 - .75}$$

$$= .651\ \text{gm m}^{-2}\ \text{yr}^{-1}$$

$$\text{or}\ \ 5969.8\ \text{kg yr}^{-1}$$

In addition, the model of Chapra (1975) could be solved using the three different values for settling velocities to provide estimates of phosphorus inputs:

$$P = \frac{L}{\nu + qs}$$

$$L = P\ (\nu + qs)$$

$$= .07\ (16 + 2.33)$$

$$= 1.283\ \text{gm m}^{-2}\ \text{yr}^{-1}$$

$$\text{or}\ \ 11766.3\ \text{kg yr}^{-1}$$

Using settling velocities of 12.4 m yr^{-1} and 10 m yr^{-1} provides loading estimates of 9455.4 kg yr^{-1} and 7914.8 kg yr^{-1} respectively.

Finally, Chapra and Tarapchak (1976) have derived a model relating chlorophyll a concentrations directly to phosphorus

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Ostrofsky, M.L. 1978. Modification of phosphorus retention models for use with lakes with low areal water loading. J. Fish. Res. Bd. Can. 35: 1532-1536.

Patalas, K. 1977. Crustacean plankton and the eutrophication of the St. Lawerence Great Lakes. J. Fish. Res. Board Can. 24:1451-1462.

Patrick, R. and C.W. Reimer. 1966. The diatoms of the United States Volume I. Monographic Series of Academy of Natural Sciences of Philadelphia #13.

Peters, R.H. 1977. Availability of atmospheric orthophosphate. J. Fish Res. Bd. Can. 34: 918-924.

Peters, R.H. 1978. Concentrations and kinetics of phosphorus fractions in water from streams entering Lake Memphremagog. J. Fish. Res. Board Can. 35: 315-328.

Philbert, F.J. 1973. A comparative study of the effect of sample preservation by freezing prior to chemical analysis of Great Lakes waters. Proc. 16th Conf. Great Lakes Res. p. 282-293.

Pieczynska, E. 1972. Production and decomposition in the eulittoral zone of lakes. Production problems of Freshwaters-Krakow 1972. Proc. IBP Unesco Symp. on Productivity Problems of Freshwaters, Kazimierz Dolny, Poland, May 6-12, 1970.

Potter, D.S. 1977. Frozen phosphates: Implications to productivity and eutrophication. Pages 295-305 in R.D. Andrews III et al. eds. Proceedings of the Symposium on Terrestrial and Aquatic Ecological Studies of the Northwest, March 26-27, 1976, Eastern Washington State College, Cheney, Washington. EWSC Press, Cheney, Washington.

Prepas, R.E., and D.O. Trew. 1983. Evaluation of the phosphorus-chlorophyll relationship for lakes off the Precambrian Shield in western Canada. Can. J. Fish. Aquat. Sci. 40:27-35.

Prescott, G.W. 1970. Algae of the Western Great Lakes Area. Wm. C. Brown Co. Pub., Dubuque, Iowa. 39 pp.

Rast, W. and G.F. Lee. 1978. Summary analysis of the North American (U.S. portion) OECD Eutrophication Project: nutrient loading-lake response relationships and trophic state indices. U.S. EPA-600/3-78-008 455 pp.

Reynolds, C.S. 1982. Phytoplankton periodicity: its motivation, mechanisms, and manipulation. Report of the Freshwater Biological Association. 50:60-75.

loadings, thereby reducing the original three steps in the modelling sequence to two. This was accomplished by combining the above equation with (a) an empirical expression relation spring phosphorus (Pv) to mean annual phosphorus (P = 0.9 Pv) and (b) with Dillon and Rigler's (1974) original phophorus-chlorophyll relationship. The condensed model is as follows:

$$\text{chl a} = 1866 \left(\frac{L}{qs + \nu}\right)^{1.449}$$

$$\text{and } L = .005\ (\text{chl } \underline{a})^{.69}\ (qs + \nu)$$

Using the observed, euphotic mean summer chlorophyll a concentration of 41.2 mg m^{-3} and an apparent settling velocity of 12.4 m yr^{-1} gives an estimated loading of:

$$
\begin{aligned}
L &= .005\ (41.2)^{.69}\ (2.33 + 12.4) \\
&= .005\ (13.0)\ (14.73) \\
&= .956\ \text{gm}\ m^{2}\ yr^{-1}
\end{aligned}
$$

or 8786.7 kg yr^{-1}

The loadings derived from these various calculations are summarized in Table 60. The range of loadings is disconcertingly large from a management viewpoint. The most obvious feature is, however, that our observed (measured) loadings were lower than those estimated by any of the theoretical techniques.

Rigler, F.H. 1968. Further observations inconsistent with the hypothesis that molybdenum blue method measures orthophosphate in lake water. Limnol. Oceanogr. 13: 7-13.

Sakamoto, M. 1966. Primary production by phytoplankton community in some Japanese lakes and its dependence on lake depth. Arch. Hydrobiol. 62: 1-28.

Saskatchewan Nelson Basin Board. 1972. Water supply for the Saskatchewan Nelson Basin. Hydrology. Appendix 4. 284 pp.

Schaffner, W.R. and Oglesby, R.T. 1978. Phosphorus loadings to lakes and some of their responses. Part II. A new calculation of phosphorus loading and its application to 13 New York lakes. Limnol. Oceanogr. 23: 120-134.

Scheider, W.A. 1978. Applicability of phosphorus budget models to small precambrian lakes, Algonquin Park, Ontario. J. Fish. Res. Board Can. 35: 300-304.

Scheider, W.A., Synder, W.R. and Clark B. 1979. Deposition of nutrients and major ions by precipitation in south-central Ontario. Water, Air and Soil Pollut. 12: 171-188.

Schindler, D.W. 1976. Biogeochemical evolution of phosphorus limitation in nutrient-enriched lakes of the Precambrian Shield. pp. 647-664. In. J.O. Nriagu [ed] Environmental Biogeochemistry. Vol. 2. Metals Transfer and Ecological Mass Balances. Ann Arbor Sci. 797 pp.

Schindler, D.W. 1978. Predictive eutrophication models. Limnol. Oceanogr. 23: 1080.

Schindler, D.W. and G.W. Comita. 1972. The dependence of primary production upon physical and chemical factors in a small senescing lake, including the effects of complete winter O_2 depletion. Arch. Hydrobiol. 69: 413-451.

Schindler, D.W., R.V. Schmidt and R.A. Reid. 1972. Acidification and bubbling as an alternative to filtration in determining phytoplankton production by the ^{14}C method. J. Fish. Res. Bd. Can. 29: 1627-1631.

Schindler, D.W., R.W. Newbury, K.G. Beaty and P. Campbell. 1976. Natural water and chemical budgets for a small precambrian lake basin in central Canada. J. Fish. Res. Bd. 33: 2526-2543.

Schindler, D.W., E.J. Fee and T. Rusczynski. 1978. Phosphorus input and its consequences for phytoplankton standing crop and production in the Experimental Lakes Area and in similar lakes. J. Fish. Res. Bd. Can. 35: 190-196.

Table 60. Phosphorus Loadings (kg yr^{-1})

Method	Loading kg yr^{-1}
Observed (1977)	3407.0
Theoretical (Calculated)	5254.0
Rast and Lee (1978) - Theoretical	4443.9
Rast and Lee (1978) - Empirical	8404.9
Schindler et al. (1978)	12028.2
Mass Balance (Short Term)	6198.0
Mass Balance (Long Term)	6169.1
Vollenweider (1969)	6566.7
Dillon & Rigler (1974)	5969.8
Chapra (1975)	11766.3
	9455.4
	7914.8
Chapra (1976)	8786.0

10.4 APPLICABILITY OF PHOSPHORUS LOADING MODELS

The preceding analysis suggests a number of obstacles to the valid application of the existing phosphorus modelling sequence to the management of Baptiste Lake, and perhaps to other similarily productive Alberta lakes.

By testing with empirical models we have shown that the annual phytoplankton standing crop is related to phosphorus concentration. In order to optimize the predictive capability of this component, improved quantitive models derived specifically for the different lake types found in this region are required.

Phosphorus concentration was, in general, poorly predicted by the steady state models; we have presented evidence to suggest that this may be partly due to the incorporation of underestimated loading data into the calculations. The determination of reliable phosphorus loading data is consequently viewed as another problem requiring study in this modelling sequence. In terms of logistics and economics, it is difficult to justify the detailed study taken on this lake for routine work. Simpler methods must be found to estimate loadings for lakes when internal phosphorus sources may be present.

There can be little doubt that the basic assumptions of steady-state models render them entirely inappropriate as management tools for stratified, eutrophic lakes such as Baptiste. Our data suggest that both external and internal loadings are highly seasonal in nature and that true steady state conditions rarely occur. Accordingly, some effort should be made towards the development and use of a "dynamic eutrophication model" when and if future lake management issues require this level of analysis.

REFERENCES

Adams, R.W. and H.C. Duthie. 1976. Relationships between sediment chemistry and postglacial production rates in a small Canadian lake. Int. Rev. ges. Hydrobiol. 61: 21-36.

Akena, A.M. 1978. An intensive surface water quality study of the Muskeg River Watershed. Vol. 1: Water Chemistry. Prep. for the Alberta Oil Sands Environmental Research Program by Alberta Environment. AOSERP Rep. 85. 187 pp.

Alberta Environment. 1976. Town of Athabasca Sewage Sampling Program, July, 1976. Municipal Engineering Branch, Pollution Control Division. Unpublished manuscript. 4 pp. & Appendix.

Alberta Environment. 1977(a). Town of Athabasca Sewage Sampling Program, August 1977. Municipal Engineering Branch, Pollution Control Division. Unpublished manuscript. 8 pp. & Appendix.

Alberta Environment. 1977(b). Alberta Surface Water Quality Objectives. Water Quality Control Branch, Standards & Approvals Division, Alberta Environment. 17 pp.

Alberta Municipal Affairs. 1977. Cottage owner survey: Island Lake, Baptiste Lake. Regional Planning Services Division, Alberta Municipal Affairs. 78 pp.

Alexander, G.G. and R.J. Stevens. 1976. Per capita phosphorus loading from domestic sewage. Wat. Res. 10: 757-764.

American Public Health Association. 1975. Standard methods for examination of water and wastewater, 14th Ed. APHA.

Anderson, J.M. 1974(a). Nitrogen and phosphorus budgets and the role of sediments in six shallow Danish lakes. Arch Hydrobiol. 74: 428-550.

Anderson, R.S. and D.K. Krochak. 1972. Report on a preliminary survey of the distribution and abundance of coliform bacteria in natural waters of some Western Canadian National Parks. Canadian Wildlife Service Manuscript Report. 60 pp.

Anderson, R.S. 1974(b). Diurnal primary production patterns in seven lakes and ponds in Alberta, Canada. Oecologia (Berl.) 14: 1-17.

Atmospheric Environmental Service, Fisheries and Environment Canada. Monthly Record. Meteorological Observation in Canada.

Atlas of Alberta. 1969. University of Alberta Press, Edmonton. 158 pp.

Baker, R.L. 1979. Birth rate of planktonic rotifers in relation to food concentration in a shallow, eutrophic lake in Western Canada. Can. J. Zool 57(6) : 1206-1214.

Bargh, B.J. 1977. Output of water, suspended sediment, and phosphorus and nitrogen forms from a small forested catchment. New Zealand J. Forestry Sci. 7: 162-171.

Bargh, B.J. 1978. Output of water, suspended sediment, and phosphorus and nitrogen forms from a small agricultural catchment. N.Z.J. Agricultural Res. 21: 29-58.

Bellinger, E.G. 1974. A note on the use of algal size in estimates of population standing crops. Br. Phycol. J. 9: 157-161.

Benson, M.A. 1965. Spurious correlation in hydraulics and hydrology. Proc. Am. Soc. Eng. HY 4:35-42.

Borneuf, D. 1972. Hydrogeology of the Tawatinaw area. Alberta. A.R.C. Report 72-11.

Bothe, R.A. 1978. Hydrological assessment of Baptiste Lake. Internal memorandum from Hydrology Branch, Technical Services Division, Alberta Environment. May 7, 1978. 5 pp.

Bradbury, J.P. 1975. Diatom stratigraphy and human settlement in Minnesota. The Geological Society of America, Special Paper 171: 1-74.

Bradbury, J.P. and J.C.B. Waddington. 1973. The impact of European settlement on Shagawa Lake, Northeastern Minnesota, U.S.A. Pages 289-307 in H.J.B. Birks and R.G. West, eds. Quaternary Plant Ecology. Blackwells, Oxford.

Brakke, D.F. 1977. Rainwater: Nutrient additions to a hypereutrophic lake. Hydrobiologia 52: 159-163.

Brydges, T.G. 1974. Chlorophyll a - total phosphorus relationships in Lake Erie. Proc. 14th Conf. Great Lakes Res: 185-190.

Caiazza, R., K.D. Hage and D. Gallup. 1978. Wet and dry deposition of nutrients in central Alberta. Water, Air, and Soil Poll. 9: 309-314.

Canada Dept. of Forestry. 1965. The C.L.I. soil capability classification for agriculture. Queen's Printer, Ottawa.

Carlson, R.E. 1977. A trophic state index for lakes. Limnol. Oceanogr. 22: 361-369.

Carignan, R. and R. J. Flett. 1981. Post depositional mobility of phosphorus in lake sediments. Limnol. Oceanogr. 26: 361-366.

Chapra, S.C. 1975. "Comment on 'An empirical method of estimating the retention of phosphorus in lakes' by W.B. Kircher and P.J. Dillon". Water Resources Res. 11: 1033.

Chapra, S.C. 1975. Comment on "An empirical method of estimating the retention of phosphorus in lakes" by W.B. Kirchner and P.J. Dillon. Water Resour. Res. 11(6):1033-1034.

Chapra, S.C. and Tarapchak, S.J. 1976. A chlorophyll a model and its relationship to phosphorus loading plots for lakes. Water Resourc. Res. 12: 1260-1264.

Chymko, N. 1977. Seasonal patterns and abundance of crustacean zooplankton in Lesser Slave Lake, Alberta, 1976. Report prepared by Fish and Wildlife Div., Alberta Recreation, Parks and Wildlife.

Cross, P.M. 1977. An attempt to estimate phosphorus retention in Bob Lake from iron retention and the iron: phosphorus ratio in the sediments. M.Sc. Thesis, University of Toronto, Toronto 121 pp.

Crowe, A.S. 1979. Chemical and hydrological simulation of prairie lake-watershed systems. M.S. Thesis, University of Alberta, Edmonton, Alta., Canada. 205 pp.

Crowe, A.S. and F.W. Schwartz. 1981(a). Simulation of lake-watershed systems. Description and sensitivity analysis of the model. J. Hydrology 52: 71-105.

Crowe, A.S. and F.W. Schwartz. 1981(b). Simulation of lake-watershed systems, II. Application to Baptiste Lake, Alberta, Canada. J. Hydrol. 52: 107-125.

Cummins, K.W. 1974. Structure and function of stream ecosystems. Bioscience 24: 621-642.

Davis, S.N. and R.J.M. Dewiest. 1966. Hydrogeology. John Wiley and Sons, New York.

Dillon, P.J. 1974. A critical review of Vollenweider's nutrient budget model and other related models. Water Res. Bull. 10: 969-991.

Dillon, P.J. and Rigler, F.H. 1974. The phosphorus-chlorophyll relationship in lakes. Limnol. Oceanogr. 19: 767-773.

Dillon, P.J. 1974. The prediction of phosphorus and chlorophyll concentrations in lakes. Ph.D. Thesis, University of Toronto, Toronto. 330 pp.

Dillon, P.J. and F.H. Rigler. 1975. A simple method for predicting the capacity of a lake for development based on lake trophic status. J. Fish. Res. Bd. Can. 32(9): 1519-1531.

Dillon, P.J. and W.B. Kirchner. 1975. The effects of geology and land use on the export of phosphorus from watersheds. Water Res. 9: 135-148.

Edmonson, W.T. 1972. Nutrients and phytoplankton in Lake Washington. In Nutrients and Eutrophication (Edited by Likens, G.E.) pp. 172-179. Limnol. Oceanogr. Spec. Symp. No. 1.

Elliot, J.M. 1977. Some methods for the statistical analysis of samples of benthic invertebrates. Freshwater Biological Association, Scientific Publication No. 25.

Fee, E.J. 1978. A procedure for improving estimates of in situ primary production at low irradiances with an incubator technique. Verh. Internat. Verein. Limnol. 20:59-67.

Fee, E.J. 1980. Important factors for estimating annual phytoplankton production in the Experimental Lakes Area. Can. J. Fish. Aquat. Sci. 37:513-522.

Feller, M.C. & Kimmins, J.P. 1979. Chemical characteristics of small streams near Haney in southwestern British Columbia. Water Resources Res. 15: 247-258.

Feth, J.H. 1961. Effects of rainfall and geology on the chemical composition of water in the coastal streams in California. Prof. Pap. 424-B 202-04. U.S. Geol. Surv. Reston, Va.

Fisher, S.G. and Likens, G.E. 1973. Energy flow in Bear Brook, New Hampshire: An integrative approach to stream ecosystem metabolism. Ecol. Monogr. 43: 421-439.

Fritz, P. and H.R. Krouse. 1973. Wabamun Lake past and present. An isotope study of the water budget. Proc. Symp. on Lakes of Western Canada, 1973, Edmonton, Alberta, Univ. Alta. Water Resour. Cent., Edmonton, Alta. Publ. No. 2. pp. 244-258.

Ganf, G.G. 1972. The regulation of net primary production in Lake George, Uganda, East Africa. Productivity Problems of Freshwaters (Ed. by Z. Kajak & A. Hillbricht. Ilkowska) pp. 693-708. PWN Polish Scientific Publishers, Warsaw-Krakow.

Ganf, G.G. 1974. Incident solar irradiance and underwater light penetration as factors controlling the chlorophyll a content of a shallow equatorial lake (Lake George, Uganda). J. Ecol. 62: 593-609.

Ganf, G.G. 1974. Photosynthetic production and irradiance-photosynthesis relationships of the phytoplankton from a shallow equatorial lake (Lake George, Uganda). Oecologia (Berl.) 18: 165-183.

Gannon, J.E. and R.S. Stemberger. 1978. Zooplankton (especially crustean and rotifers) as indicators of water quality. Trans. Amer. Micros. Soc. 97: 16-35.

Gasith, A. and Lawacz, W. 1976. Breakdown of leaf litter in the littoral zone of a eutrophic lake. Ekol. Pol. 24: 421-430.

Gasith, A. and Hasler, A.D. 1976. Airborne litter fall as a source of organic matter in lakes. Limnol. Oceanogr. 21: 253-258.

Gelin, C. 1975. Nutrients, biomass and primary productivity of nanoplankton in eutrophic Lake Vombsjon, Sweden. Oikos 26: 121-139.

Golterman, H.L. 1975. Physiological limnology, an approach to the physiology of lake ecosystems. Elsevier Scientific Publishing, New York. 489 pp.

Gomolka, R.E. 1975. An investigation of atmospheric phosphorus as a source of lake nutrients. M. Sc. Thesis, Univ. of Toronto. 167 pp.

Gray, D.M. 1973. Handbook on Principles of Hydrology. Water Information Center Inc., Washington, 504 pp.

Hallegraeff, G.M. 1976. Pigment diversity, biomass and species diversity of phytoplankton of three Dutch lakes. Ph. D. Thesis, Univ. of Amsterdam.

Hallegraeff, G.M.. 1977. A comparison of different methods used for the quantitative evaluation of biomass of freshwater phytoplankton. Hydrobiol. 55: 145-165.

Hammer, U.T., K.F. Walker, and W.D. Williams. 1973. Derivation of daily phytoplankton production estimates from short-term experiments in some shallow, eutrophic Australian saline lakes. Aust. J. Mar. Freshwat. Res. 24: 259-266.

Hickman, M. 1974. Effects of the discharge of thermal effluent from a power station on Lake Wabamun, Alberta, Canada - the epipelic and epipsammic algal communities. Hydrobiolgia 45: 199-215.

Hickman, M. 1978. Ecological studies on the epipelic algal community in five prairie-parkland lakes in central Alberta. Can. J. Bot. 56: 991-1000.

Hickman, M. 1979(a). Seasonal succession, standing crop and determinants of primary productivity of the phytoplankton of Ministik Lake, Alberta, Canada. Hydrobiol. 64: 105-121.

Hickman, M. 1979(b). Phytoplankton of shallow lakes: Seasonal sucession, standing crop and the chief determinants of primary productivity 1. Cooking Lake, Alberta, Canada. Holarctic Ecol. 1: 337-350.

Hickman, M. and C.G. Jenkerson. 1978. Phytoplankton primary production and population efficiency studies in a prairie-parkland lake near Edmonton, Alberta, Canada. Int. Rev. ges. Hydrobiol. 31: 1-25.

Hickman, M., D.M. Klarer, C. Schweger and T. Habgood. 1978. The paleoenvironmental history of Baptiste Lake, Alberta. A report submitted to Alberta Environment, Water Quality Control Branch, 45 pp. (Excerpts in text).

Hillman, T.J. 1975. Comparison of materials for enclosure of samples during carbon-14 measurements of phytoplankton productivity. Bull. Aust. Soc. Limnol. 6:1-4.

Holdren, G.C. and D.E. Armstrong. 1980. Factors affecting phosphorus release from intact sediment cores. Environ. Sci. Technol. 14: 79-87.

Holm-Hansen, O., C. Lorenzen, R.W. Holmes, and J.D.H. Strickland. 1965. Fluorometric determination of chlorophyll. J. Cons. Perm. Int. Explor. Mer. 30:3-15.

Hustedt, F. 1949. Susswasser-Fiatomeen aus dem Albert National Park in Belgisch-Congo. Explor. Parc Natn. Albert Misc. H. Damas (1935-36) 8: 1-99.

Hutchinson, G.E. 1967. A Treatise of Limnology, V. II. Introduction to Lake Biology and the Limnoplankton. John Wiley & Sons, N.Y. 1115 pp.

Jackson, M.B. 1976. Water Quality in the Bay of Quinte prior to phosphorus removal at sewage treatment plants. Ontario Ministry of Environment Toronto, Ont. 62 pp.

Jansson, M. 1979. Nutrient-budget and regulation of nutrient concentrations in a small sub-arctic lake in northern Sweden. Freshwat. Biol. 9: 213-231.

Javornicky, P. and Kormarkova. 1973. The changes in several parameters of phytoplankton productivity in Slapy Reservoir 1960-1967, their mutual correlations and correlations with the main ecological factors. Hydrobiol. Stud. Prague 2: 155-211.

Jewson, D.H. 1977. Light penetration in relation to phytoplankton content of the euphotic zone of Lough Neagh, N. Ireland. Oikos 28: 74-83.

Johnson, A.H., D.R. Bouldin, E.A. Govette and A.M. Hedges. 1976. Phosphorus loss by stream transport from a rural watershed: Quantities, processes and sources. J. Environ. Qual. 5: 148-156.

Johnson, N.M., Likens, G.E., Bormann, F.H. Fisher, D.W., and Pierce, R.S. 1969. A working model for the variation in streamwater chemistry at the Hubbard Brook Experimental Forest, New Hampshire. Water Resour. Res. 5: 1353-1363.

Jones, R.A. and G.F. Lee. 1977. Septic tank wastewater disposal systems as phosphorus sources for surface waters. Occasional paper 13, Center for Env. Studies, University of Texas, Dallas, Richardson. 72 pp.

Jones, J.R. and Bachman, R.W. 1976. Prediction of phosphorus and chlorophyll levels in lakes. J. Water Pollut. Contr. Fed. 48: 2176-2182.

Kaushik, N.K., J.B. Robinson, P. Sain, H.R. Whitney, and W. Stammers. 1975. A quantitative study of nitrogen losses from water of a small spring-fed stream. Proc. 10th Canadian Symp. Water Poll. Res. Canada: 110-117.

Kaushik, N.H. and J.B. Robinson, 1976. Preliminary observations on nitrogen transport during summer in a small spring-fed Ontario stream. Hydrobiol. 49: 59-63.

Kirchner, W.B. and Dillon P.J. 1974. An empirical method of estimating the retention of phosphorus in lakes. Water Resources Res. 11: 182-183.

Kjearsgaard. 1972. Soil survey of the Tawatinaw Map Sheet (831): Alberta Soils Survey No. 29.

Landers, D. 1982. Effects of naturally senescing aquatic macrophytes on nutrient chemistry and chlorophyll a of surrounding waters. Limnol. Oceanogr. 27: 428-439.

Larsen, D.P. and Mercier H.T. 1976. Phosphorus retention capacity of lakes. J. Fish Res. Bd. Can. 33: 1742-1750.

Laycock, A.H. 1968. Water balance of the Gull Lake Basin, a system analysis for the period 1921-1968. Rep. for Red Deer Reg. Planning Comm., Alta. Environ. Edmonton, Alta.

Laycock, A.H. 1971. Water balance for the Cooking Lake region. Presented to Publ. Hear. Environ. Conserv. Auth. Cooking Lake, Alta. Environ., Edmonton, Alta.

Laycock, A.H. 1973. Lake level fluctuations and climatic variation in central Alberta. Proc. Symp. on Lakes of Western Canada, 1973, Edmonton, Alta. Univ. Alta. Water Resour. Cent., Publ. No. 2, pp. 38-98.

Lean, D.R.S. 1973. Movement of phosphorus between its biologically important forms in lake water. J. Fish. Res. Bd. Canada. 30(10): 1525-1536.

Lee, G.F. 1973. Role of phosphorus in eutrophication and diffuse source control. Wat. Res. 7: 111-128.

Lee, G.F. Bentley, E. and R. Amundson. 1975. Effects of marshes on water quality. Chapter 4. pp. 105-127. Springer-Verlag. New York.

Lehman, J.T. 1975. Reconstructing the rate of accumulation of lake sediment; the effect of sediment focusing. Quaternary Res. 5: 541-550.

Likens, G. 1975. Drinary production in inland aquatic ecosystems. pp. 185-202 in: Primary Productivity of the Biosphere. H. Leith and R. Whittaker. eds. Springer-Verlag, New York.

Likens, G.E., F.H. Bormann, R.S. Pierce, J.S. Eaton and N.N. Johnson. 1977. Biogeochemistry of a Forested Ecosystem. Springer-Verlag, New York. 146 pp.

Livingstone, D.A. and J.C. Boykin. 1962. Vertical distribution of phosphorus in Linsey Pond Mud. Limnol. Oceanogr. 7: 57-62.

Long, E.T., and G.D. Cooke. 1978. Phosphorus variability in three streams during storm events: chemical analysis vs. algal assay. Mitt. Internat. Verein. Limnol. 21:441-452.

Lund, J.W.G. 1970. Primary production. Water Treat. Exam. 19:332-358.

Lund, J.W.G., F.J.H. Mackereth, and C.H. Mortimer. 1963. Changes in depth and time of certain chemical and physical conditions and of the standing crop of Asterionella formosa Hass. in the north basin of Windermere in 1947. Phil. Trans. Roy. Soc. London [Ser. B] 246: 255-290.

MacKereth, F.J.H. 1966. Some chemical observations on post glacial lake sediments. Phil. Trans. Royal Soc. London 250 (B): 165-213.

Mathews, C.P., and A. Kowalezewski. 1969. Disappearance of leaf litter and its contribution to production in the River Thames. J. Ecol. 57:543-552.

McCart, P.J. et al. 1979. Limnological and fishery surveys of the aquatic ecosystems at Esso Resources' Cold Lake Lease. Aquatic Environment Ltd. Calgary, Alberta. 267 pp.

McColl, R.H.S. 1972. Chemistry and trophic status of seven New Zealand lakes. N.Z.J. Mar. Freshwater Res. 6: 399-477.

Megard, R.O. 1969. Algae and photosynthesis in Shagawa Lake, Minnesota. Minnesota Univ. Limnol. Res. Centre Interim Rep. 5, 20 pp.

Megard, R.O. 1972. Phytoplankton, photosynthesis, and phosphorus in Lake Minnetonka, Minnesota. Limnol. Oceanogr. 17:68-87.

Megard, R.O., W.S. Combs, P.D. Smith, and A.S. Knoll. 1979. Attenuation of light and daily integral rates of photosynthesis attained by plankton algae. Limnol. Oceanogr. 24:1038-1050.

Meyer, J.L. 1979. The role of sediments and bryophytes in phosphorus dynamics in a headwater stream ecosystem. Limnol. Oceanogr. 24: 365-375.

Miller, R.B. 1944. Suggestions for experiments to control the Pike-Whitefish tapeworm, Triaenophorus crassus. Alberta Dept. Lands and Forsts. Ms. Rep.

Miller, R.B. and H.B. Watkins. 1946. An experiment in the control of the cestode Triaenophorus crassus forel. Canadian J. Research, D. 24: 175-179.

Minns, C.K. and Johnson, M.G. 1979. Temporal variation in the nutrient export of rivers draining into the Bay of Quinte, Ontario, Canada. Water Resources. Bull. 15: 1061-1072.

Mitchell, P. 1979. Skeleton, Garner, Muriel Lakes water quality studies. Water Quality Control Branch, Pollution Control Div., Alta. Env. 106 pp.

Mortimer, C.H. 1941. The exchange of dissolved substances between mud and water in lakes. Parts III and IV: summary and references. J. Ecol. 30: 147-201.

Mortimer, C.H. 1971. Chemical exhanges between sediments and waters in the Great Lakes - speculations on probable regulatory mechanisms. Limnol. and Oceanogr. 16: 387--404.

Murphy, T.P., and B.G. Brownlee. 1981. Ammonia volatilization in a hypertrophic prairie lake. Can. J. Fish. Aquat. Sci. 38:1035-1039.

National Eutrophication Survey. 1974. The relationships of phosphorus and nitrogen to the trophic state of northeast and north-central lakes and reservoirs. Working paper No. 23. N.E.S., Pacific N.W. Environmental Research Lab., Corvalis, Oregon. 28 pp.

Nicholls, K.H. 1976. Nutrient-phytoplankton relationships in the Holland Marsh, Ontario. Ecol. Monogr. 46: 179-199.

Nicholls, K.H. 1977. The phytoplankton of the Kawartha Lakes (1972 and 1976). pp. 29-45. Chapter 2 of the Kawartha Lakes Water Management Study Ont. Min. Envir., Ont. Min. Natural Res., Toronto, Canada. 185 pp.

Nicholls, K.H. and C.M. Cox. 1978. Atmospheric nitrogen and phosphorus loading to Harp Lake, Ontario, Canada. Water Resources Res. 14: 589-592.

Nicholls, K.H. and D.J. Dillon. 1978. An evaluation of phosphorus-chlorophyll-phytoplankton relationships for lakes. Int. Revue-ges. Hydrobiol 63: 141-154.

Noton, L.R. 1974. Phytoplankton productivity in Lake Wabamun, Alberta, and the effect of thermal effluent. M.Sc. Thesis, University of Alberta, Edmonton. 163 pp.

Nursall, J.R., J.B. Nuttal, and P. Fritz. 1971. The effect of thermal effluent on Lake Wabamun. XVIII Limnological Congress, Leningrad.

Nusch, E.A. 1975. Comparative investigations on extent, causes and effects of eutrophication in western German reservoirs. Verh. Int. Verein. Limnol. 19: 1871-1879.

Nygaard, G. 1977. New or interesting plankton algae, with a contribution on their ecology. Der Kongelige Danske Videnskabernes Selskab Biologiske Skrifter. 21: 1-107.

O.E.C.D. 1981. Co-operative program on monitoring of inland waters (eutrophication control), synthesis report. R.A. Vollenweider and J.J. Kerekes, eds. Paris, France.

Oglesby, R.T., W.R. Schaffner, and E.L. Mills. 1975. Nitrogen, phosphorus and eutrophication in the Finger Lakes. Tech. Rep. No. 94. Cornell Univ. Water Resources and Marine Sci. Center, Ithaca, N.Y. 27 p.

Omernik, J.M. 1977. Non-point source stream nutrient level relationships. A nation-wide study. E.P.A. 600/5-77-105 Corvallis Environ. Res. Lab., Environ. Protection Agency, Corvallis, Ore. 151 pp.

Ongley, E.D., J. Ralston, and R.L. Thomas. 1977. Sediment and nutrient loadings to Lake Ontario: methodological arguments. Can. J. Earth Sci. 14: 1555-1565.

Schoemann, F.R. 1973. A systematic and ecological study of the diatom flora of Lesotho with special reference to water quality. V&R Printers. Pretoria. 355 pp.

Schwartz, F.W. and D.N. Gallup. 1978. Some factors controlling the major ion chemistry of small lakes, examples from the prairie parkland of Canada. Hydrobiologia 58: 65-81.

Sharpley, A.N., J.K Syers, and P.W. O'Connor. 1976. Phosphorus inputs into a stream draining an agricultural watershed. 1. Sampling. Water, Air and Soil Pollut. 6: 39-52.

Siegel, S. 1956. Nonparametric Statistics for the Social Sciences McGraw-Hill, Toronto. 311 p.

Smith, G.M. 1950. The fresh-water algae of the United States. McGraw-Hill Book Co., New York.

Smith, V.H. 1979. Nutrient dependence of primary productivity in lakes. Limnol. Oceanogr. 24: 1051-1064.

Soeder, C.J. and J.F. Talling. 1971. The enclosure of phytoplankton communities. In: A manual on methods for measuring primary production in aquatic environments. (Edited by R.A. Vollenweider), p 62-70. IBP Handbook No. 12. Blackwell, Oxford.

Sonzogni, W.C. and G.F. Lee. 1974. Nutrient sources for Lake Mendota-1972. Trans. Wis. Acad. Sci. Arts. Lett. 62: 133-164.

Steeman Neilsen, E. 1952. The use of radioactive carbon (^{14}C) for measuring organic production in the sea. J. Cons. int. Explor. de la Mer 18: 117-140.

Stein, J.R. 1975. Freshwater algae of British Columbia: the Lower Frazer Valley. Syesis. 8: 199-184.

Stone, D. 1970. The process of rural settlements in the Athabasca Area, Alberta. M.A. Thesis, Dept. of Geography, University of Alberta, Edmonton. 189 pp.

Strickland, J.D.H. and T.R. Parsons. 1972. A practical handbook of seawater analysis. 2nd edition. Fisheries Research Board of Canada, Ottawa. 310 pp.

Szczepanski, A. 1965. Deciduous leaves as a source of organic matter in lakes. Bull. de L'academie Polanaise des Sciences. 13:215-217.

Taft, C.E. and C.W. Taft. 1971. The algae of western lake Erie. Bull. Ohio Biol. Surv. 4: 1-189.

Talling, J.F. 1960. Self-shading effects in natural populations of a planktonic diatom. Wett. Leben. 12: 235-242.

Talling, J.F. 1965. The photosynthetic activity of phytoplankton in East African Lakes. Int. Rev. Ges. Hydrobiol. 50: 1-32.

Talling, J.F., R. B. Wood, M. V. Prosser, and R. M. Baxter. 1973. The upper limit of photosynthetic productivity by phytoplankton: evidence from Ethiopian soda lakes. Freshwat. Biol. 3:53-76.

Theodorsson, P. and J. Bjarnason. 1975. The acid-bubbling method for primary productivity measurements modified and tested. 20: 1018-1022.

Trew, D.O., D.J. Beliveau and E.I. Yonge. 1978. Baptiste Lake study summary report. Water Quality Control Branch, Pollution Control Div., Alta. Environ. Edmonton, Alta. 105 pp.

Trew, D.O., E.I. Yonge, and R.P. Kaminski. 1981. A trophic assessment of nine lakes in the Cold Lake region. Water Quality Control Branch, Pollution Control Div., Alta. Environ. Edmonton, Alta. 70 pp.

Tsui, T.P., R. Green and G.J. Mann. 1978. Winter limnological studies of the Cold Lake Project Area. An interim report to Imperial Oil Limited, Department of New Energy Resources, Calgary, Alberta. 93 pp.

U.S. EPA. 1974. Technology transfer report on methods for chemical analysis of water and waste. EPA. 625/6-74-003. p. 249-255.

Uttormark, P.D., J.D. Chapin and K.M. Green. 1974. Estimating nutrient loading of lakes from non-point sources. U.S. EPA Report EPA-600/3-74-020. 122 pp. and nutrient recycling. Arch. Hydrobiol. 80: 40-69.

Vollenweider, R.A. 1968. The scientific basis of lake and stream eutrophication, with particular reference to phosphorus and nitrogen as eutrophication factors. Technical Report OECD, Paris, DAS/DS1/ 68, 27: 1-182.

Vollenweider, R.A. 1969. Moglichkeiten und grenzen elementarer modelle der stoffbilanz Von Seen. Arch. Hydrobiol. 65:1-36.

Vollenweider, R.A. 1975. Input-output models with special reference to the phosphorus loading concept in limnology. Schwiez. Z. Hydrol. 37: 58-84. 1975.

Ward, R.C. 1975. Principles of hydrology. McGraw-Hill publishers.

Water Survey of Canada. Hydrometric Field Procedures, Manual III. Unpublished Manuscript. 83 pp.

Watson, N.H.F. and G.F. Carpenter. 1974. Seasonal abundance of Crustean zooplankton and net phytoplankton biomass of Lakes Huron, Erie and Ontario. J. Fish. Res. Board Can. 31: 309-317.

Webber, C.I. 1971. A guide to the common diatoms at water pollution surveillance system stations. U.S. Environmental Protection Agency, National Environmental Research Centre, Analytical Quality Control Laboratory, Cincinnati, Ohio 45268.

Wetzel, R.G. 1975. Limnology. W.B. Saunders Company Philadelphia. 743 pp.

Widger, W.K. and K.D. Kimball. 1976. Comments on "An empirical method of estimating the retention of phosphorus in lakes: by W.B. Kirchner and P.J. Dillon. Water Resources Res. 12: 122.

Winberg, G.G., ed. 1971. Methods for the estimation of production of aquatic animals. Academic Press, New York. 175 pp.

Winberg, G.G. 1971. Symbols, units, and conversion factors in studies of freshwater productivity. IBP, Cable Printing Servies, London.

Wood, B.J.B., P.B. Tett, and A. Edwards. 1973. An introduction to the phytoplankton, primary production and relevant hydrography of Loch Etive. J. Ecol. 61: 560-585.

Wright, H.E., E.J. Cushing and D.A. Livingstone. 1975. Coring devices for Lake sediments. Pages 494-520 in B. Kummel and D. Raup, eds. Handbook of Paleontological Techniques W.H. Freeman, San Francisco.

Yentsch, C.S. and Menzel D.W. 1963. A method for determination of phytoplankton phaeophytin by fluorescence. Deep Sea Res. 10: 221-231.

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Table A1. Chlorophyll a at Baptiste north (mg m^{-3})

Date	0 m	0.5 m	1.0 m	1.5 m	2.0 m	3.0 m	10.0 m
1976							
Mar 16		34.0	34.0		5.0	2.0	1.1
Mar 30			14.0		13.4	9.0	0.2
Apr 28		27.2	27.8		35.0	24.2	8.3
May 05		20.5	23.4		19.4	16.0	10.8
May 19		8.3	10.1		9.7	8.8	7.8
May 26		4.8	5.7		5.7	5.8	4.7
Jun 01		7.2	7.0		7.6	8.1	7.4
Jun 08		25.1	25.7		24.8	20.0	3.5
Jun 15		22.8	21.7		22.5	21.4	1.5
Jun 22		68.9	66.3		57.6	49.7	3.8
Jun 28		57.4	60.0		44.0	39.3	6.4
Jul 08		78.5	72.0		55.0	20.9	1.8
Jul 14		30.0	38.3		37.5	22.8	0.8
Jul 20		63.8	58.9		62.8	57.9	0.8
Aug 02		71.5	75.6		33.4	20.3	4.0
Aug 11		660.7	84.4		68.7	32.8	6.6
Aug 24		38.0	64.1		72.0	75.9	2.2
Aug 30		840.5	126.9		206.7	111.2	10.6
Sep 08		49.7	48.4		47.1	42.3	47.1
Sep 15		44.5	60.2		57.6	47.1	43.2
Sep 21		140.4	82.4		53.6	47.1	11.4
Sep 27		38.0	40.4		37.5	27.1	5.5
Oct 05		23.8	20.4		16.4	29.9	7.5
Oct 20		22.3	22.8		24.6	23.8	24.9
Oct 27		14.9	15.2		15.8	15.5	16.0
Nov 03		10.3	10.6		10.0	10.0	10.3
Dec 01		3.9	3.3		3.0	2.0	1.5
Dec 08		1.4	1.3		1.3	1.1	0.6
Dec 15		1.3	0.9		0.6	0.5	0.4
Dec 22	2.2	1.4	0.5	0.3	0.5	0.3	0.3

Table A1. Chlorophyll a at Baptiste north (mg m^{-3})

Date	0 m	0.5 m	1.0 m	1.5 m	2.0 m	3.0 m	10.0 m
1977							
Jan 05	1.4	1.0	0.8	0.6	0.5	0.4	0.1
Jan 19	0.8	0.7	0.4	0.2	0.4	0.1	0.1
Feb 02	1.1	0.9	0.5	0.5	0.2	0.2	0.1
Feb 16	0.9	0.8	0.1	0.4	0.1	0.4	0.3
Mar 09	2.1	2.4	2.0	1.0	0.8	0.3	0.1
Mar 23	3.7	5.1	2.3	3.5	2.1	1.8	0.2
Apr 28	16.2	14.5	20.5	15.4	14.5	13.3	7.1
Mar 11	9.0	13.3	11.1	10.8	11.8	11.5	6.4
Mar 18	10.5	10.5	11.8	11.5	12.4	14.5	7.1
May 25	10.0	10.0	11.9	12.1	11.6	18.3	1.0
May 31	7.8	8.1	8.8	15.2	15.7	10.2	6.7
Jun 22	16.9	16.7	14.8	16.2	17.0	12.2	0.9
Jul 06	62.7	60.6	56.5	59.1	63.4	47.7	14.9
Jul 13	48.5	46.3	54.9	62.0	50.6	43.5	9.0
Jul 19	70.7	68.7	72.6	74.6	70.7	64.8	2.7
Jul 27	19.0	60.2	49.7		73.3		3.3
Aug 03	52.3	6.1	7.4	24.9	19.0	7.2	2.8
Aug 10	89.0	89.0	66.5	38.0	60.8	74.1	31.4
Aug 17	157.0	125.6	45.1	107.9	117.8	45.6	7.8
Aug 24	23.6	33.8	29.1	26.7	26.7	25.5	3.2
Aug 31	43.5	44.5	44.2	42.0	42.8	40.6	42.8
Sep 08	41.9	52.3	56.3	56.3	52.3	57.6	16.6
Sep 13	57.6	57.6	58.9	55.0	57.6	58.9	57.6
Sep 21	53.6	55.0	48.4	52.3	48.4	52.3	37.9
Sep 28	43.2	38.5	39.2	38.5	39.9	31.4	31.4
Oct 05	32.8	25.7	33.3	30.9	30.4	29.0	42.8
Oct 12	27.1	24.7	24.7	28.5	24.2	26.1	28.0
Oct 17	21.9	17.1	21.9	18.1	17.6	18.5	21.9
Oct 26	16.8	17.0	17.3	16.0	17.3	16.0	12.6
Nov 02	13.6	16.2	12.8	14.4	15.2	14.4	15.4
Nov 09	10.7	12.6	13.6	16.5	13.9	16.0	15.4

Table A2. Chlorophyll a at Baptiste south (mg m^{-3})

Date	0 m	0.5 m	1.0 m	1.5 m	2.0 m	3.0 m	10.0 m	20.0 m
1976								
Mar 16		1.8	1.1		0.9	0.8	0.2	0.1
Mar 30			7.8		12.5	8.3	1.2	0.8
Apr 28		29.6	21.7		19.9	11.6	3.6	1.1
May 05		30.8	29.0		27.8	27.8	13.4	3.9
May 17		14.0	16.2		14.7	15.9	16.0	2.4
May 19		18.2	16.7		12.5	11.4	13.8	12.3
May 26		7.9	8.0		8.0	7.9	7.2	1.3
Jun 01		10.3	11.8	11.0	10.7	11.0	4.4	2.1
Jun 08		25.9	25.9		51.8	28.3	30.6	1.1
Jun 15		34.5	32.2		30.6	28.3	1.8	0.7
Jun 22		49.7	50.6		45.4	41.9	1.6	0.4
Jun 28		64.4	38.5		12.0	9.9	1.6	0.9
Jul 07		111.2	90.3		39.4	8.5	1.1	0.4
Jul 14		27.1	28.5		27.1	27.1	1.2	0.3
Jul 20		41.2	41.2		34.3	29.6	1.0	0.3
Jul 28		20.7	26.0		27.4	26.4	0.5	0.3
Aug 02		41.2	34.3		37.3	39.3	12.8	12.6
Aug 11		70.7	64.8		68.7	55.6	2.6	1.7
Aug 17		16.8	38.5			35.6	2.5	
Aug 24		55.0	49.7		53.6	53.6	4.7	2.8
Aug 30		117.8	89.0		77.2	66.7	15.0	1.5
Sep 08		39.4	51.0		52.3	47.1	2.3	
Sep 15		33.7	35.2		31.8	31.8	24.7	14.7
Sep 21		41.3	38.0		30.9	21.9	0.6	1.4
Sep 29		38.5	29.5		29.5	27.1	3.2	1.6
Oct 05		29.6	28.5		40.2	44.2	44.0	16.4
Oct 13		19.6	18.5		19.7	19.0	20.0	3.0
Oct 20		18.9	16.0		17.9	17.6	18.5	16.2
Oct 25		25.3	23.4		25.3	25.3	23.8	20.5
Nov 03		17.1	16.8		16.2	17.7	17.1	18.2
Nov 10		9.1	7.9		7.4	7.1	8.3	7.9
Dec 15		2.8	0.8		0.6	1.1	0.5	0.7

Table A2. Chlorophyll a at Baptiste south (mg m^{-3})

Date	0 m	0.5 m	1.0 m	1.5 m	2.0 m	3.0 m	10.0 m	20.0 m
1977								
Jan 05	1.2	1.2	1.0	0.7	0.6	0.6	0.3	0.2
Jan 19	1.2	0.9	0.3	0.4	0.6	0.5	0.1	0.1
Feb 02	1.5	1.4	1.0	0.8	0.6	0.2	0.2	0.1
Feb 16	1.3	1.1	1.1	0.5	0.4	0.2	0.1	0.2
Mar 09	1.1	0.1	1.0	1.3	1.1	0.7	0.1	0.1
Mar 23	22.9	16.2	4.2	3.1	3.1	1.3	0.1	0.1
Apr 27	16.2	12.6	13.1	13.5	13.3	10.0	9.0	8.1
May 11	13.1	15.3	19.6	22.9	17.3	12.8	9.3	2.4
May 18	18.3	16.9	14.3	16.2	15.4	14.7	10.2	3.0
May 25	22.9	16.2	4.2	3.1	2.1	1.3	0.1	0.1
May 31	14.1	14.1	13.7	15.1	15.4	13.4	2.8	1.4
Jun 08	61.8	28.5	22.1	19.2	14.7	13.7	1.5	1.0
Jun 22	13.1	12.8	11.8	11.8	10.5	10.9	0.9	0.5
Jul 06	32.8	39.2	32.1	32.8	31.4	23.2	1.4	1.2
Jul 13	57.0	49.2	47.0	52.7	51.3	37.8	1.9	0.5
Jul 18	22.4	21.6	22.0	22.8	23.6	23.6	2.2	
Jul 27	47.1	75.9	51.0	44.5	35.2	19.5	6.5	0.6
Aug 03	35.3	41.2	39.9	39.3	44.5	32.1	1.6	0.2
Aug 10	27.6	53.2	75.1	76.0	70.3	40.9	0.8	1.1
Aug 17	235.5	227.7	208.0	176.6	104.0	84.4	1.7	0.6
Aug 24	74.6	66.7	78.5	74.6	74.6	76.5	3.2	1.7
Aug 31	18.1	20.8	22.0	13.4	19.6	21.2	6.7	1.4
Sep 08	41.9	45.8	44.5	45.8	44.5	31.4	9.6	1.6
Sep 13	30.9	28.0	33.7	29.9	29.9	32.8	11.5	2.5
Sep 21	20.4	18.3	19.6	16.7	19.4	19.1	10.2	2.4
Sep 28	34.9	38.5	36.3	30.6	20.8	31.4	20.0	14.9
Oct 05	9.7	9.7	9.4	9.4	8.6	8.4	4.4	5.1
Oct 12	8.9	8.4	8.6	9.0	8.4	8.6	7.9	2.0
Oct 17	5.5	5.7	5.5	4.9	4.7	5.9	6.1	6.2
Oct 26	5.9	6.4	6.0	5.5	6.1	5.4	5.8	5.0
Nov 02	5.9	5.6	6.0	6.4	6.6	6.7	6.3	6.3
Nov 09	6.9	5.9	7.4	6.1	6.9	7.1	5.9	6.7
Nov 16	8.6	7.7	6.1	5.9	6.7	7.9	6.7	7.1
Dec 13	6.4	6.7	6.6	5.7	5.1	4.7	2.6	4.3

Table A3. Chlorophyll a in Ghost Lake (mg m^{-3})

Date	0 m	0.5 m	1.0 m	1.5 m	2.0 m	3.0 m	6.0 m	10.0 m
1976								
Mar 30		3.3	0.7		0.8	1.0		1.0
Apr 29			2.6		3.0	3.0		1.2
Jun 02		4.1	5.1		4.6	4.6		1.5
Jun 30		9.6	9.1		8.6	7.9		2.9
Jul 26		13.0	12.0		9.0	13.0		1.8
Sep 01		6.9	11.3		7.1	9.9		4.1
Sep 29		11.4	12.1		12.0	12.3		13.9
Nov 01		15.2	14.7		21.9	7.7		10.7
Dec 20	1.0	1.0	1.5	1.0	1.0	1.3		1.8
1977								
Jan 26	0.5	0.4	0.3	0.2	0.1	0.1		0.2
Feb 22	1.6	3.4	1.7	2.7	1.3	.5		1.0
Mar 30	36.0	11.4	6.2	4.0	2.4	1.5	.7	1.1
Apr 29	9.8	9.5	10.7	11.6	11.4	11.2	10.8	4.3
May 26	7.8	19.5	15.9	18.3	12.6	13.5		2.8
Jul 25	20.6	21.3	20.6	26.2	33.4	30.1	6.2	2.7
Aug 29	7.9	9.2	7.8	8.6	8.1	7.0	7.9	5.8
Oct 04	8.9	7.9	10.2	8.6	9.7	8.3	9.2	9.5
Nov 08	25.7	21.4	18.5	20.4	18.1	21.4	17.1	24.9

Table A4. Chlorophyll a in Island Lake (mg m^{-3})

Date	0 m	0.5 m	1.0 m	1.5 m	2.0 m	3.0 m	6.0 m	10.0 m
1976								
Mar 31			1.7		1.5	1.7		1.1
Apr 29		1.5	2.1		2.3	1.5		0.4
Jun 09		2.2	2.0		2.5	2.8		0.1
Jun 30		2.1	2.1		3.1	3.4		5.1
Jul 26		1.6	1.8		1.9	1.6		0.8
Sep 01		6.5	6.4			5.8		-ve
Sep 29		9.1	8.9		9.0	7.2		3.7
Nov 01		10.7	16.6		17.6	6.1		10.2
1977								
Feb 07	1.3	1.7	2.5	1.1	1.3	1.0		0.8
Feb 15	4.5	1.2	2.1	1.2	1.1	1.1		2.6
Apr 05	1.6	0.9	1.0	0.7	0.7	0.7	1.6	0.6
Apr 29	7.6	5.9	7.0	7.5	7.4	10.1	4.6	2.4
May 24	2.5	1.9	1.8	1.4	1.4	1.2		1.7
Jul 21	2.4	2.3	2.7	2.5	2.6	2.8		4.5
Aug 29	5.1	4.7	3.9	4.9	4.4	5.0	5.0	0.1
Sep 27	7.8	7.8	9.7	9.5	9.0	10.0	9.8	-ve

Table A5. Phaeophytin a at Baptiste north (mg m^{-3})

Date	0 m	0.5 m	1.0 m	1.5 m	2.0 m	3.0 m	6.0 m	10.0 m
1976								
Apr 28		5.1	6.7		6.6	3.4		3.0
May 05		3.4	3.1		3.4	3.4		3.6
May 17		3.6						
May 19		3.6	3.2		4.3	3.9		2.8
May 26		1.7	1.8		2.0	1.9		2.3
Jun 01		2.8	2.1	2.7	2.5	3.0	2.5	2.4
Jun 08		2.3	2.3		4.3	2.8		4.5
Jun 15		2.6	3.0		3.2	2.7		1.9
Jun 22		4.5	2.7		5.0	2.4		1.7
Jun 28		6.5	4.7		6.6	6.2		2.0
Jul 07		6.0	2.7		4.0	3.2		1.6
Jul 14		1.2	4.2		3.3	1.5		1.3
Jul 20		4.8	5.2		4.9	3.5		1.6
Jul 28								
Aug 02		4.2	2.1		3.7		2.6	1.1
Aug 11		4.3	7.7		5.4		1.1	0.7
Aug 17								
Aug 24		4.0	6.9		6.3		10.8	0.4
Sep 08		5.7	3.4		3.5		3.2	3.5
Sep 15		6.1	4.8		3.8	3.5		4.8
Sep 21		6.7	7.9		5.3	3.5		0.2
Sep 29		2.6	3.3		4.0	3.1		1.2
Oct 05		4.2	3.2		3.1	8.8		1.5
Oct 13								
Oct 20		3.5	3.0		3.3	4.2		2.8
Oct 25		3.2	2.9		3.4	2.8		2.9
Nov 03		2.9	2.8		2.9	2.9		2.9
Dec 01		0.9	0.6		0.6	0.6	0.6	0.7
Dec 08		1.0	0.9		0.5	0.8		1.1
Dec 15		0.5	0.4		0.6	0.5		0.7

Table A5. Phaeophytin a at Baptiste north (mg m-$_3$)

Date	0 m	0.5 m	1.0 m	1.5 m	2.0 m	3.0 m	6.0 m	10.0 m
1977								
Jan 05	0.2	0.3	0.3	0.3	0.3	0.3		0.5
Jan 19	0.2	0.2	0.1	0.1	0.1	0.2		0.3
Feb 02								
Feb 16	0.4	0.5	0.3		0.5	0.2		0.2
Mar 09	0.6	0.7	0.6	1.4	1.3	0.2	0.3	0.3
Mar 23	0.9	0.5	0.6	0.7	0.6	0.7		0.2
Apr 27	2.6	3.0	2.0	1.8	2.8	2.4	4.1	6.4
May 11	0.3	1.6	0.3	0.3	0.8	1.9		1.2
May 18	0.4	0.1		1.8	1.3	1.9		2.1
May 25	1.0	1.4	0.6	1.7	2.7	1.4		0.3
May 31	0.1	0.5	0.6	1.2		1.6		4.0
Jun 22	0	0	0	0	0	0		0.7
Jul 06	0	0	0	0	0	0		1.3
Jul 13	0	0	0	0	0	0		3.0
Jul 18	0	0	0	0	0	0		0.8
Jul 27	9.9	0	0	1.4		2.0		1.1
Aug 03	0	0	4.2	0	4.3	6.1	3.9	0.8
Aug 10	-	0.1	-	-	-	-	-	-
Aug 17	1.9	-	-	2.2	1.6	1.6	2.1	2.0
Aug 24	1.4	1.9	0.6	1.9	0.8	1.9	2.2	1.3
Aug 31	1.1	1.6	0.4	2.5	1.8	2.7	2.7	1.8
Sep 08	5.1	1.8	0.3	-	3.0	0.2	3.0	2.2
Sep 13	3.9	3.9	5.6	4.2	3.9	5.6	7.9	3.9
Sep 21	1.7	0.4	2.2	3.0	2.4	0.6	3.3	7.0
Sep 28	3.8	2.2	3.4	3.5	3.4	4.1	2.2	0.4
Oct 05	1.8	2.8	0.9	2.8	2.4	2.5	1.8	0.8
Oct 12	1.3	2.0	2.4	2.5	3.3	1.8	2.1	1.7
Oct 17	3.1	2.1	1.3	3.0	3.0	2.9	0	1.8
Oct 26	2.5	2.8	2.2	2.6	2.0	2.1	2.4	2.4
Nov 02	2.3	1.6	2.4	1.5	2.2	2.0	0	2.9
Nov 09	1.1	0.7	1.6	1.1	1.1	1.1	1.0	1.4

Table A6. Phaeophytin a at Baptiste south (mg m^{-3})

Date	0 m	0.5 m	1.0 m	1.5 m	2.0 m	3.0 m	6.0 m	10.0 m	20.0 m
1976									
Mar 16		0.6	0.3		0.2	0.2		0.2	0.2
Mar 30			5.7		6.1	4.6		0.6	0.4
Apr 28		2.7	7.1		3.4	3.5		1.9	0.4
May 05		7.0	5.5		5.5	6.7		4.6	1.9
May 17		5.1	4.9		4.6	4.7		6.2	0.5
May 19		4.0	5.1		3.6	3.3		4.1	2.5
May 26		2.9	2.8		3.3	2.8		1.2	1.6
Jun 01		3.1	2.7	3.1	3.1	3.4	3.4	3.4	1.1
Jun 08		1.7	2.7		2.5	2.6		1.1	0.6
Jun 15		4.5	3.9		2.6	3.5		1.9	0.7
Jun 22		3.2	4.0		2.8	3.1		1.4	0.5
Jun 28		1.5	2.7		6.6	2.1		1.1	2.6
Jul 07		4.4	8.6		2.5	1.9		1.1	0.4
Jul 14		1.8	1.3			1.1		0.6	0.5
Jul 20		3.0	3.0		3.6	3.5		0.7	0.5
Jul 28		2.6	1.9		1.7	2.2		0.5	0.4
Aug 02		3.0	1.8		2.4	3.2		2.3	1.2
Aug 11		1.9	7.4		5.8	5.3		0.7	0.5
Aug 17		3.2	3.5		3.6	3.7		1.0	0.6
Aug 24		5.0	4.5		5.3	7.8		1.0	0.7
Aug 30		6.2	7.3		6.1	6.7		1.7	0.6
Sep 08		8.2	5.6		5.4	4.7		1.0	
Sep 15		2.6	2.4		4.0	2.7		3.3	0.5
Sep 21		2.4	3.0		3.7	3.1		1.3	0.5
Sep 29		2.2	2.9		2.9	4.0		1.5	0.9
Oct 05		2.9	2.4		7.6	4.6		7.1	2.7
Oct 13		1.6	2.5		1.5	3.7		2.2	0.5
Oct 20		1.1	1.3		2.7	2.1		2.0	1.8
Oct 25		1.5	1.3		1.9	1.9		1.6	2.0
Nov 03		2.0	1.5		1.6	1.7		2.0	1.9
Nov 10		1.4	1.4		1.3	1.2		1.5	1.5
Dec 10		0.2	0.2		0.3	0.3	0.3	0.3	0.3

Table A6. Phaeophytin *a* at Baptiste south (mg m^{-3})

Date	0 m	0.5 m	1.0 m	1.5 m	2.0 m	3.0 m	6.0 m	10.0 m	20.0 m
1977									
Jan 05	0.8	0	0.3	0.2	0.2	0.2		0.1	0.3
Jan 19	0.1	0	0	0.1	0.1	0.1		0.1	0.2
Feb 02									
Feb 16	0.3	0.3	0.2	0.2	0.1	0.2		0.1	0.2
Mar 09	0.4	0.4	0.4	0.3	0.4	0.3	0.2	0.2	0.4
Mar 23	12.1	5.7	1.9	1.9	1.0	0.7		0.2	0.2
Apr 27	1.8	2.1	2.7	2.1	2.7	3.2	3.2	2.4	2.4
May 11	2.6	2.1	3.2	2.4	2.3	1.1		0.6	0.6
May 18	1.8	1.3	1.9	0.9	1.6	1.9		1.8	1.0
May 25	0.7	2.7	1.6	0.3	1.1	4.1		2.1	1.2
May 31	-	-	-	-	0.9	-		0.9	1.0
Jun 08	0	0	0	0	0	0	0	1.1	0.5
Jun 22	0	0	0	0	0	0	0	0.7	0.3
Jul 06	0	0	0.7	0	0	0	0	0.7	1.3
Jul 13	0	0	0	0	0	0	0	0.1	0.3
Jul 18	0	0	0	0	0	0	0	2.2	
Jul 27	0	0	0	0	1.1	1.1	8.2	2.5	0.3
Aug 03	0	0	0	0	0	0	0.8	0.9	0.3
Aug 10	3.0	0	0	0	0	0	2.0	1.2	0.3
Aug 17	0	0	0	0	0	0	0	1.0	0.4
Aug 24	1.3	0	0.9	1.3	0	0	0	1.0	0.2
Aug 31	3.6	2.0	1.1	1.2	1.3	1.6	0.7	1.3	0.3
Sep 08	3.9	1.2	2.5	2.4	6.1	1.0	2.1	1.9	0.3
Sep 13	1.9	1.3	1.6	1.5	1.1	0	1.3	2.1	0
Sep 21	0	1.9	1.3	1.3	1.8	0.9	1.9	1.8	0.5
Sep 28	1.1	1.5	1.0	2.1	5.2	1.8	1.3	2.4	1.0
Oct 05	1.6	1.2	0.9	1.4	1.7	1.3	1.2	1.1	0.6
Oct 12	0.8	1.4	0.5	1.7	2.1	1.7	1.5	1.3	0.5
Oct 17	1.0	0.8	1.1	0.7	0.9	0.8	0.6	1.2	0.9
Oct 26	0.8	0.9	1.0	1.1	0.9	1.1	1.0	0.9	0.7
Nov 02	1.1	1.0	0.7	0.9	0.8	0.9	0.8	1.0	0.9
Nov 09	0	0.2	0.1	0.4	0.3	0.1	0.3	0.5	0.3
Nov 16	0	0.5	0.8	0.4	0.6	0.1	0.3	0.6	0.7

Table A7. Phaeophytin in Ghost Lake (mg m^{-3})

Date	0 m	0.5 m	1.0 m	1.5 m	2.0 m	3.0 m	6.0 m	8.0 m	10.0 m
1976									
Mar 31		0.8	0.8		0.9	0.7			0.8
Apr 29		3.5	3.5		0.8	0.8		1.4	
Jun 02		1.5	1.1		1.8	1.7			1.9
Jun 30		1.8	1.6		4.1	1.0			3.7
Jul 26	0.6	0.6	0.9		2.0	0.9			2.2
Sep 01	1.3	1.3	1.9		0.8	1.4			1.4
Sep 29	2.5	2.5	2.6		2.2	2.2			2.5
Nov 01	3.2	3.2	4.5		7.5	2.9			3.7
Dec 20	0.6	0.8	0.7	0.6	0.7	0.6	0.9		
1977									
Jan 26	0.2	0.1	0.1	0.1	0.1	0.1		0.2	
Feb 22	0.7	0.1	0.1	0.2	0.2	0.7		1.8	
Mar 30	0	1.1	0.8	1.2	1.2	0.8	1.5	1.5	8.0
Apr 29	0.4	0.1	0.7	0.4	1.1	0.9	0.8	1.9	1.7
May 26	0.5	0.2	0.9	0	0.7	1.8			2.4
Jul 25	0.8	0.7	1.4	0	0.3	1.2	1.3		2.1
Aug 29	0	0.2	0	0.2	0	0	0	0.7	
Oct 04	0	1.5	0	1.5	0.2	0	0.2	0.4	
Nov 08	0	0	1.8	0	0	0.3	0	0	3.2

Table A8. Phaeophytin _a_ in Island Lake (mg m^{-3})

Date	0 m	0.5 m	1.0 m	1.5 m	2.0 m	3.0 m	6.0 m	10.0 m
1976								
Mar 31	1.0	1.0	1.0		1.1	1.4		0.8
Apr 29	1.7	1.7	1.6		1.5	1.3		2.5
Jun 09	0.3	0.3	0.2		0.6	0.6		0.7
Jun 30	0.4	0.4	0.6		0.7			1.3
Jul 26	0.2	0.2	0.6		0.4	0.8		8.1
Sep 01	0.4	0.4	0.6			0.5		1.2
Sep 29	1.4	1.4	1.3		1.2	1.3		5.7
Nov 01	1.9	1.9	4.4		3.8	1.3		2.1
1977								
Feb 07	0.2	0.3	0.1	0.4	0.5	0.4		0.9
Feb 15	0.5	0.3	0.4	0.4	0.4	0.4		0.8
Apr 05	1.7	0.9	0.7	0.4	0.4	0.4	0.4	1.8
Apr 29	1.6	1.7	2.1	1.8	1.6	1.3	1.1	1.6
May 24	0.1	0.2	0.2		0.2	0.4		0.2
Jul 21	0.1	0.2	0.8	0.6	0.8	0.6		1.8
Aug 29	0.1		0.2	0.3	0.8	0.2		0.2
Sep 27	1.6	2.0	1.2	1.9	0.7	2.3	2.1	

Table A9. Cell counts at Baptiste north (cells mL^{-1})

Date	0 m	0.5 m	1.0 m	1.5 m	2.0 m	3.0 m	6.0 m	10.0 m
1976								
Mar 16			34,595		1,943	345		609
Mar 30			16,471		8,426	5,030		2,579
Apr 28		24,426	46,263		23,336	14,165		13,847
May 05		5,811	8,390		9,134	5,865		2,887
May 19		8,444	8,916		8,935	16,653		8,836
May 26		2,579	2,216		1,562	1,380		2,131
Jun 01		41,638	16,376		19,126	17,656		21,293
Jun 08		70,007	75,636		57,930	45,400		1,217
Jun 15		37,183	36,456		51,756	52,664		4,604
Jun 22		184,052	140,513		134,202	121,536		15,036
Jun 28		90,573	149,593		122,353	206,797		12,894
Jul 07		245,701	197,081		135,065	43,402		1,761
Jul 14		102,377	119,765		107,053	103,557		288
Jul 20		108,915	105,010		72,685	57,386		845
Aug 02		186,291	134,656		43,902	18,372		3,647
Aug 11	10,846,060		222,642		183,961	159,263		3,186
Aug 24		76,772	60,791		75,636	98,700		3,382
Aug 30	13,887,860		247,468		258,009	78,454		14,747
Sep 08	82,900	65,558	65,467		62,652	37,228		32,052
Sep 15	49,468	80,585	57,749		67,646	56,387		110,821
Sep 21	667,198	528,214	244,434		274,922	190,498		66,481
Sep 27	203,846	86,487	99,350		54,715	45,087		9,550
Oct 05	37,174	16,326	31,980		29,256	57,004		18,995
Oct 20	11,183	14,029	18,432		12,909	17,618		15,466
Oct 27	11,978	13,635	11,742		11,068	12,589		5,769
Nov 03	8,315	7,956	6,627		7,076	7,176		7,461
Dec 01		6,091	3,859		2,672	3,346		2,226
Dec 08		3,248	1,567		1,638	2,090		1,589
Dec 15		2,930	1,690		1,751	1,387		1,354
Dec 20	1,255	1,905	1,093		731	949		2,468

Table A9. Cell counts at Baptiste north (cells mL^{-1})

Date	0 m	0.5 m	1.0 m	1.5 m	2.0 m	3.0 m	6.0 m	10.0 m
1977								
Jan 05	3,193	3,829	1,930	1,420	1,577	2,562		685
Jan 19	3,675	2,134	1,172	1,022	3,827	628		1,667
Feb 03	1,086	3,565	1,860	1,105	3,844	471		6,430
Feb 16	916	650	437	600	618	507		5,902
Mar 09	2,670	2,670	1,489	417	560	81	3,018	9,608
Mar 23	1,559	1,324	1,440	6,702	1,585	3,043		5,609
Apr 28	22,564	9,307	11,305	13,438	10,356	9,965	9,625	17,933
May 11	17,252	20,839	19,068	14,483	13,938	19,749		7,014
May 18	8,467	7,536	7,355	9,035	8,967	8,740		5,266
May 25	15,481	10,699	11,622	11,728	12,258	52,165		3,233
May 31	12,333	9,489	13,211	18,281		10,260		3,178
Jun 22	15,860	24,993	37,364	40,928	42,699	24,039		12,107
Jul 06	102,649	73,412	115,089	79,859	82,900	99,335		45,082
Jul 13	115,952	104,284	99,244	81,947	109,369	90,437		5,614
Jul 19	149,820	145,552	154,541	155,313	79,995	149,094		3,889
Jul 27	115,407	144,827	99,109	87,085	105,010	32,007	7,884	9,037
Aug 03	33,460	56,561	77,861	28,239	129,526	43,543	13,475	4,853
Aug 10	92,117	83,264	49,486	71,505	95,249	98,336	35,639	78,951
Aug 17	194,630	92,661	140,649	103,875	103,058	33,051	7,552	13,136
Aug 24	45,355	48,919	25,742	35,140	28,782	10,896	65,694	4,374
Aug 31	38,545	69,825	63,878	19,386	25,288	72,640	54,934	49,259
Sep 08	88,167	28,375	40,497	47,852	31,553	98,155	30,645	20,589
Sep 13	67,215	50,939	64,771	80,842	62,652	37,561	80,086	92,828
Sep 21	76,681	70,824	68,826	42,177	41,768	78,678	51,938	83,037
Sep 28	116,224	90,759	109,323	46,944	41,496	53,345	80,585	22,064
Oct 05	28,244	67,918	56,106	28,451	39,876	35,261	44,719	60,745
Oct 12	20,839	28,216	22,972	13,944	43,062	26,894	31,840	42,608
Oct 17	26,695	26,962	19,991	17,076	41,102	36,320	9,962	14,301
Oct 26	21,701	3,042	7,355	14,165	11,160	18,054	24,955	25,227
Nov 02	32,461	4,812	19,595	51,120	5,766	12,273	10,031	21,186
Nov 09	16,193	33,672	15,905	25,355	21,943	21,111	18,811	8,323

Table A10 . Cell counts at Baptiste south (cells mL^{-1})

Date	0 m	0.5 m	1.0 m	2.0 m	3.0 m	10.0 m	20.0 m
1976							
Mar 16		672	272	272	288	54	648
Mar 30			13,787	12,092	11,496	1,453	1,150
Apr 28		34,413	33,006	18,659	17,751	10,851	24,068
May 05		31,671	26,695	58,521	23,109	13,302	8,354
May 17		7,427	12,821	9,225		11,524	9,643
May 19		17,833	11,840	16,108	11,404	6,574	12,785
May 26		2,415	11,205	2,088	1,453	4,410	43,039
Jun 01		17,430	24,819	14,015	17,462		39,089
Jun 08		82,265	49,077	74,047	101,333	71,596	60,700
Jun 15		82,401	83,672	48,760	53,209	4,507	109,823
Jun 22		88,258	130,253	178,013	153,679	1,162	1,144
Jun 28		204,754	148,912	40,633	43,039	3,323	588
Jul 07		813,009	322,794	138,288	18,569	1,142	2,456
Jul 17		77,498		86,714	104,193	115,225	1,652
Jul 20		143,555	125,077	76,499	65,421	734	5,240
Jul 28		76,908	71,641	91,209	55,524	1,725	1,407
Aug 02		94,432	86,759	74,274	66,329	26,150	18,284
Aug 11		320,342	230,632	283,478	346,130	6,628	1,265
Aug 17		125,667	113,591	152,726	123,351	5,463	1,139
Aug 24		89,362	83,536	88,530	112,743	8,377	8,139
Aug 30		249,538	108,688	72,958	66,965	10,368	17,479
Sep 08	96,293	75,137	80,449	80,948	92,752	15,375	9,080
Sep 15		58,712	32,839	63,241	110,095	98,595	8,735
Sep 21	1,467,328	36,320	54,480	31,175	49,032	25,805	11,641
Sep 27	380,270	83,486	130,298	86,525	48,670	10,084	21,497
Oct 05		19,086	15,944	16,980	16,417	7,173	14,891
Oct 13		6,370	11,865	14,831	8,308	7,219	8,762
Oct 20	9,927	10,366	10,245	18,175	9,110	8,263	11,759
Oct 27	12,971	11,272	7,220	10,940	7,675	7,140	15,219
Nov 03		3,982	5,593	6,190	7,271	3,761	4,336
Nov 10		5,053	4,186	2,878	3,147	2,583	5,493
Dec 15		3,125	2,434	2,404	1,301	470	941

Table A10. Cell counts at Baptiste south (cells mL^{-1})

Date	0 m	0.5 m	1.0 m	1.5 m	2.0 m	3.0 m	10.0 m	20.0 m
1977								
Jan 05	3,755	2,300	3,208	2,629	1,949	1,659	708	2,904
Jan 19	586	798	877	742	605	395	423	5,508
Feb 02	2,050		543	1,177	454	738	504	1,142
Feb 16	1,332	4,430	417	421	1,054	100	227	266
Mar 09	3,924	2,937	1,063	1,805	420	306	136	53,045
Mar 23	54,616	28,466	6,658	2,897	1,619	1,214	483	42,131
Apr 27	11,713	9,125	10,442	11,017	11,395	10,215	11,577	14,437
May 11	22,291	25,515	26,910	22,291	22,569	17,888	11,864	12,099
May 18	14,664	14,074	17,751	16,526	15,618	10,236	9,806	34,141
May 25	12,590	11,743	16,662	14,377	14,392	9,685	5,539	25,311
May 31	23,722	24,652	18,092	17,139	12,621	13,961	1,998	19,522
Jun 08	57,204	73,730	62,879	29,782	27,512	24,652	1,930	2,581
Jun 22	30,236	29,673	24,334	49,214	23,540	39,770	610	46,442
Jul 06	50,803	42,721	65,603	45,309	60,541	37,705	1,385	20,907
Jul 13	101,651	85,897	111,412	80,767	105,283	117,904	1,585	21,656
Jul 19	46,216	62,289	58,990	48,336	49,940	56,054	4,438	18,569
Jul 27	75,942	48,169	86,669	64,967	25,515	33,184	11,532	15,482
Aug 03	52,346	43,539	23,336	36,275	24,652	19,704	3,260	9,307
Aug 10	72,640	51,166	101,832	44,946	87,667	62,970	1,755	9,549
Aug 17	208,931	230,541	218,419	388,034	114,136	71,868	5,705	6,961
Aug 24	53,572	69,627	77,316	91,617	94,023	56,069	7,633	7,824
Aug 31	78,814	25,606	42,222	22,700	27,739	16,571	4,631	7,037
Sep 08	40,451	44,946	62,788	73,185	28,829	28,103	15,368	18,478
Sep 13	45,355	14,528	84,353	54,071	15,845	16,027	28,935	6,924
Sep 21	21,111	19,023	30,282	14,324	27,149	19,658	19,999	13,915
Sep 28	80,237	26,536	21,701	18,342	19,277	14,210	34,118	28,239
Oct 05	17,393	27,650	14,490	14,543	16,019	15,813	9,088	8,725
Oct 11	7,476	8,835	4,400	7,673	12,027	15,590	23,600	9,749
Oct 17	9,007	7,789	6,154	5,902	5,086	12,028	7,170	5,514
Oct 26	7,830	7,892	7,139	14,776	4,162	2,020	2,831	6,993
Nov 02	2,300	4,661	4,722	4,623	6,556	4,159	3,384	7,884
Nov 09	3,002	3,855	7,635	2,771	2,339	3,125	5,037	1,378
Nov 16	4,210	3,749	13,142	3,900	3,349	4,509	3,257	5,599

Table A11. Cell counts in Ghost Lake (cells mL^{-1})

Date	0 m	0.5 m	1.0 m	1.5 m	2.0 m	3.0 m	4.0 m	6.0 m	8.0 m	10.0 m
1976										
Mar 30			31		236	140				18
Apr 29		89910	59383		62779	27149				9461
Jun 02		3178	3449		3559	1234				681
Jun 30		13693	17778		12331	11441				4975
Jul 26		44901	36193		43357	31907				4013
Sep 01		8735	20430		17978	35775				4177
Sep 29	4013	8461	9861		34686	12457				5321
Nov 01		5758	18145		11215	6620				5661
Dec 20	486	156	123		191	402				10572
1977										
Jan 26	562	184	45	8	77	51				357
Feb 22	336	499	421	251	218	200				889
Mar 30	7461	4828	2860	2406	1721	1258	700	316	1321	2300
Apr 29	8595	6294	11124	10397	10170	12425	10215	9141	6008	2610
May 26	8611	24788	117797	19008	10654	3693				2426
Jul 25	13865	14205	15139	16412	18225	11581		688		7021
Aug 29	11261	14374	8634	10377	6056	7033		6018		2942
Oct 04	7430	2700	2344	7782	5872	1649		1641		3216
Nov 08	2702	2442	2211	2385	2039	3470		3784		3244

Table A12 Cell counts in Island Lake (cells mL^{-1})

Date	0 m	0.5 m	1.0 m	1.5 m	2.0 m	3.0 m	6.0 m	10.0 m
1976								
Mar 31			145		156	109		236
Apr 29		3832	3613		2016	3632		7500
Jun 09		1416	7120		3532	10496		27331
Jun 30		5448	3214		2472			3432
Jul 26		981	1362		2397	2452		5212
Sep 01	63605	55161	51847		52573	31916		33278
Sep 29	22555	41479	24879		21837	31598		89847
Nov 01	5540	4508	5440		4826	3519		3536
1977								
Feb 07	1031	1755	2038	1332	1362	172		16798
Feb 15	4902	22046	5599	568	354	299		1041
Apr 05	9549	6084	3950	2373	2058	2416	1438	12982
Apr 29	7264	10094	6462	5145	6992	7385	4065	3923
May 24	1581	845	510	423	542	996		8914
Jul 21	2987	1792	2624	1735	2534	4625		34504
Aug 29	8033	5275	6886	7735	6355	11577	8126	17161
Sep 27	21004	24428	21042	24017	24562	24788	23532	22019

Table A13. Phytoplankton of Baptiste, Ghost and Island Lakes

Baptiste Lake North

Chlorophyta:	Cell Volume (um³)
Ankistrodesmus braunii (Naeg.) Brunnthaler	141
Ankistrodesmus falcatus (Corda) Ralfs.	241
Cerasterias staurastroides West & West	2,170
Closterium acerosum (Shank) Ehrenberg	1,877
Dictyosphaerium sp.	47
Micrasterias sp.	2,125
Oocystis parva West & East	151
Pandorina sp.	824
Pediastrum duplex Meyen	162
Scenedesmus acuminatus (Lag) Chodat	117
Scenedesmus arcuatus Lemmerman	77
Scenedesmus quadricauda (Trup) de Brebisson	63
Tetraedron minimum (A. Br.) Hansgirg	135
Unidentified green flagellate	113
Cyanophyta:	
Anabaena circinalis Rabenhorst	53
Anabaena sp.	419
Aphanizomenon flos-aquae (Lyngb.) de Brebisson	75
Gloeocapsa sp.	524
Gomphosphaeria naegelianum Lemmerman	42
Lyngbya sp.	57
Merismopedia tenuissima Lemmerman	14
Microcystis sp.	14
Spirulina sp.	4
Unidentified filamentous blue-green	2
Unidentified unicellular blue-green	48
Chrysophyta:	
Dinobryon divergens Imhof	625
Mallomonas sp.	2,669
Bacillariophyta:	
Asterionella formosa Hass.	364
Cocconeis placentala Ehrenberg	312
Fragilaria capucina Desm.	216
Gomphonema sp.	702
Melosira granulata (Ehrenberg) Ralfs.	335
Rhizosolenia eriensis H.L. Smith	1,375
Stephanodiscus astreae (Ehrenberg) Grun.	3,803 - 27,720
Stephanodiscus hantzschii Grun.	250
Synedra sp.	232

Pyrrhophyta:

Ceratium hirundinella (O.F. Muell) Dujardin	67,169
Peridinium sp.	6,605

Cryptophyta:

Cryptomonas ovata Ehrenberg	1,413
Rhodomonas minuta Skuja	105

Baptiste Lake South

Chlorophyta:

Actinastrum sp.	64
Ankistrodesmus falcatus (Naeg.) Brunnthaler	241
Cerasterias staurastroides West & East	2,170
Closterium sp.	85
Closterium acerosum (Shank) Ehrenberg	1,877
Cosmarium sp.	1,877
Dictyosphaerium sp.	42
Micrasterias sp.	2,125
Oocystis parva West & East	151
Pandorina sp.	180 - 824
Pediastrum duplex Meyen	162
Scenedesmus acuminatus (Lag) Chodat	117
Scenedesmus arcuatus Lemmerman	77
Scenedesmus quadricauda (Turp) de Brebisson	63
Selenastrum sp.	20
Tetraedron minimum (A. Br.) Hansgrig	135
Tetrastrum elegans Playfair	14

Cyanophyta:

Anabaena circinalis Rabenhorst	53
Aphanizomenon flos-aquae (Lyngb.) de Brebisson	75
Aphanocapsa sp.	33
Gomphosphaeria naegelianum Lemmerman	42
Merismopedia tenuissima Lemmerman	14
Spirulina sp.	4
Unidentified filamentous blue-green	3
Unidentified unicellular blue-green	48

Chrysophyta:

Dinobryon divergens Imhof.	625
Mallomonas sp.	2,699

Bacillariophyta:

Asterionella formosa Hass	364
Cocconeis placentula Ehrenberg.	312
Fragilaria capucina Desm.	177 - 216

Fragilaria construens (Ehr.) Grun	177
Melosira granulata (Ehrenberg) Ralfs.	335
Stephanodiscus astraea (Ehrenberg) Grun.	3,803 - 27,720
Stephanodiscus hantzschii Grun	250
Synedra sp.	232

Pyrrhophyta:

Ceratium hirundinella (O.F. Muell) Dujardin	67,169
Peridinium sp.	33,524

Cryptophyta:

Cryptomonas ovata Ehrenberg	1,413
Rhodomonas minuta Skuja	105

Ghost Lake

Chlorophyta:

Ankistrodesmus braunii (Naeg.) Brunnthaler	141
Ankistrodesmus falcatus (Corda) Ralfs.	241
Cerasterias straurastroides West & East	2,170
Closterium acerosum (Schrank) Ehr	1,877
Dictyosphaerium sp.	47
Oocystis parva West & East	151
Pandorina sp.	824
Pediastrum duplex Meyen	162
Scenedesmus quadricauda (Turp.) de Breibisson	63
Tetraedron minumum (A.Br.) Hansgrig	135
Unidentified Green Flayellate	524

Cyanophyta

Anabaena circinalis Rabenhorst	53
Anabaena sp.	419
Aphanizomenon flos-aquae (Lynzg. de Breibisson)	75
Gomphosphaeria naegelianum Lemmerman	42
Lyngbya sp.	1,037
Merismopedia tenuissima Lemmerman	14
Oscillatoria sp.	37
Unidentified filamentous blue-green	2
Unidentified unicellular blue-green	14

Chrysophyta

Dinobryon divergens Imhof.	625

Bacillariophyta

Asterionella formosa Hass.	364
Fragilaria capucina Desm.	369
Fragilaria crotonensis (Ehrenberg) Grun.	801

Melosira granulata (Ehrenberg) Grun.	335
Stephanodiscus astraea (Ehrenberg) Grun.	19,982 - 22,691
Synedra sp.	232

Pyrrhophyta:

Ceratium hirundinella (O.F. Muell) Dujardin	67,169
Peridinium sp.	5,413

Cryptophyta:

Rhodomonas minuta Skuj.	105

Island Lake

Chlorophyta:

Actinastrum sp.	180
Ankistrodesmus braunii (Naeg.) Brunnthaler	117
Ankistrodesmus falcatus (Corda) Ralfs.	241
Closterium acerosum (Schrnak) Ehrenberg	1,877
Dictyosphaerium sp.	47
Oocystis parva West & West	151
Pandorina sp.	824
Pediastrum boryanum (Turp) Meneghini	162
Peridinium sp.	5,413
Scenedesmus quadricauda (Turp) de Brebisson	63
Selenastrum sp.	20 - 101
Tetraedron muticum (A.Br.) Hansgrig	135
Tetrastrum elegans Playfair	14
Unidentified unicellular Green	824

Cyanophyta:

Anabaena circinalis Rabenhorst	53
Aphanizomenon flos-aquae (Lyngb) de Brebisson	75
Aphanocapsa sp.	14
Gomphosphaeria naegelianum Lemmerman	42
Lyngbya sp.	1,037
Merismopedia tenuissima Lemmerman	14
Oscillatoria sp.	25
Spirulina sp.	4
Unidentified filamentous blue-green	2
Unidentified unicellular blue-green	14

Chrysophyta:

Dinobryon divergens Imhof	625
Mallomonas sp.	2,112

Bacillariophyta:

Asterionella formosa Hass.	364
Cocconeis placentula Ehrenberg	312
Fragilaria crotonensis (Ehrenberg) Grun.	801
Melosira granulata (Ehrenberg) Grun.	335
Rhizosolenia eriensis H.L. Smith	1,375
Stephanodiscus astraea (Ehrenberg) Grun.	3,290 - 13,867
Synedra sp.	232

Pyrrhophyta:

Ceratium hirundinella (O.R. Muell) Dujardin	67,169

Cryptophyta:

Cryptomonas ovata Ehrenberg	1,413
Rhodomonas minuta Skuj	105

Table A14. Phytoplankton primary production at Baptiste north (mg C m$_{-3}$ hr$_{-1}$)

Date	0 m	0.5 m	1.0 m	1.5 m	2.0 m	3.0 m	6.0 m	10.0 m
1976								
Oct 20		19.6	22.7		21.0	8.0		2.2
Oct 27		7.8	4.1		2.4	1.4		1.8
Nov 03		8.2	8.0		7.7	9.9		4.6
Dec 15	0.6	0.8	0	0.1	0.3	0		0
Dec 22	0.4	0.2	0.4	0.2	0	0.4		0
1977								
Jan 05	0.5		0.3	0.4	0.4	0.1		0
Jan 19	0.4	0.6	0.6	0.2	0.3	0.3		0
Feb 02	1.1	0.1	0.7	1.3	0	0		0
Feb 16	1.3	1.6	1.3	0.7	0.5	0.1		0
Mar 09	2.4	2.7	2.7	0.9	1.9	0		0.2
Mar 23	3.1	3.9	1.3	1.4	0.9	0.9		0.9
Apr 27	12.1		26.8	21.3	13.1	6.2		0.2
May 31	17.4	17.5	25.4	24.0	18.6	11.1		1.5
Jul 18	203.9	281.9	227.0	116.1	44.0	6.3		0.8
Jul 27	54.5	116.1	93.6	27.5	9.1	2.4	1.3	0.3
Aug 10	249.4	242.0	115.2	38.5	32.2	7.4	3.2	0.5
Aug 17	292.7	207.9	5.7	7.8	2.2	1.1	1.2	2.1
Aug 24	24.5	25.8	22.1	11.2	4.9	5.2	0.7	0
Aug 31	63.1	67.1	18.4	28.1	15.8	2.1	1.3	0.7
Sep 08	109.6	134.1	111.0	75.4	44.9	0	1.3	1.6
Sep 13	115.0	141.0	116.8	79.9	47.2	10.1	2.8	4.1
Sep 21	117.1	78.8	57.3	24.8	11.7	4.7	1.7	3.4
Sep 28	99.4	83.3	67.8	37.5	24.8	9.4	4.7	3.5
Oct 05	70.0	68.9	56.8	37.6	18.0	7.2	1.5	1.1
Oct 12	62.2	61.4	55.8	46.1	24.1	9.8	0.3	0
Oct 17	39.1	40.6	37.7	26.8	14.9	6.3	1.4	1.6
Oct 26	16.5	19.8	19.4	23.3	19.4	13.6	1.5	0.8
Nov 02	8.0	2.4	5.7	5.8	1.1	2.3	1.4	0.2
Nov 09	8.2	13.5	13.7	14.9	10.9	6.5	4.2	1.3

Table A16. Phytoplankton primary production in Ghost Lake (mg C m^{-3} hr^{-1})

Date	0 m	0.5 m	1.0 m	1.5 m	2.0 m	3.0 m	6.0 m	10.0 m
1976								
Sep 29		38.6	38.8		25.2	12.6		1.1
Dec 20	0.7	0.4	0.7	0.7	0.6	0.2		0.7
Jan 26	1.1	0.5	0.9	0.4	0.6	0.7		0.3
Feb 22	0.1	0.2	0.5	0.1		0.2		0.1
1977								
Mar 30	18.2	2.4	1.2	0.4	0.3	0.7		0.5
Apr 29	6.1	12.9	24.8	22.1	10.8	6.2		0.1
May 26	13.3	39.2	38.1	45.2	19.8	16.2		0.5
Jul 25	86.9	79.2	76.6	68.8	60.0	32.4	2.6	0
Aug 29	15.8	13.5	13.3	11.2	7.5	5.9	0.3	0.2
Oct 04	11.4	16.2	14.5	11.9	8.3	6.1	0.2	0.3
Nov 08	12.5	14.8	15.3	16.7	12.2	4.2	0	

Table A18. Dark bottle uptake at Baptiste north (mg C m^{-3} hr^{-1})

Date	0 m	0.5 m	1.0 m	1.5 m	2.0 m	3.0 m	6.0 m	10.0 m
1976								
Nov 03	2.2	2.2	1.2		2.3	5.8		4.3
Dec 15	2.0	2.0	2.4	2.1	2.2	2.7		2.2
Dec 22	1.1	1.8	1.6	1.3	1.9	0.8		1.2
1977								
Jan 05	0.8	2.1	0.7	0.4	0.4	0.8		0.9
Jan 19	0.7	0.7	0.6	0.9	0.9	0.7		1.4
Feb 02	0.8	0.9	0.9	0.9	1.9	2.5		1.9
Feb 16	0.7	0.9	1.3	0.8	0.8	1.2		0.7
Mar 09	1.9	2.0	2.0	2.3	1.8	3.2		3.7
Mar 23	3.1	2.9	2.7	3.2	1.7	2.2		
Apr 28	2.0		1.8	1.8	4.8	2.0		0.8
May 31	1.3	1.4	1.7	1.4	0.6	1.2		0.7
Jul 06	-	-	-	-	-	-	-	-
Jul 18	5.6	3.6	7.7	6.1	2.3	4.9		0.8
Jul 27	3.6	5.7	5.9	4.8	5.0	3.9	1.7	4.7
Aug 10	7.3	9.7	6.0	3.7	2.2	2.2	0.8	1.1
Aug 17	10.1	5.8	8.0	5.5	6.7	5.4	1.4	1.5
Aug 24	2.1	2.7	2.5	2.2	2.4	2.1	2.0	2.3
Aug 31	-	-	-	-	-	-	-	-
Sep 08	5.0	5.7	5.1	12.7	4.1	7.3		2.2
Sep 13	2.5	1.7	2.9	2.0	4.1	2.7	2.8	1.9
Sep 21	4.1	1.7	1.4	1.6	3.0	5.1	4.2	1.8
Sep 28	2.4	1.8	2.4	5.4	2.4	2.4	0.8	1.4
Oct 05	1.3	6.0	1.7	3.1	1.7	3.4	2.3	2.3
Oct 12	1.7	1.1	1.1	0.8	1.0	0.7	0.8	1.4
Oct 17	1.8	1.2	0.8	1.0	1.5	1.2	0.4	0.6
Oct 26	0.6	0.6	4.1	1.2	1.3	0.5	0.9	0.7
Nov 02	3.1	6.2	1.6	1.6	3.5	1.2	1.4	1.3
Nov 09	0.1	0.1	1.4	0.4	0.9	0.1	0.1	1.3

Table A19. Dark bottle uptake at Baptiste south (mg C m^{-3} hr^{-1})

Date	0 m	0.5 m	1.0 m	1.5 m	2.0 m	3.0 m	6.0 m	10.0 m	20.0 m
1976									
Nov 03		1.3	2.4		1.7	4.7		6.1	2.9
Nov 10	2.0	6.8	0.7	1.8	1.8	3.0	0.4	1.1	1.2
Dec 15	3.5	3.8	3.7	3.5					
1977									
Jan 05	0.7	2.1	1.6	2.1	1.7				
Jan 19	1.5	1.7	0.8	1.8	1.9	2.0		1.7	
Feb 16	1.6	1.4	2.1		2.0	1.8		1.9	1.4
Mar 09	1.0	1.4	1.8	1.7	1.9	1.5		2.9	2.4
Mar 23	1.4	1.5	2.0	2.5	2.3	2.4		2.9	2.6
Apr 28	1.8	1.6	1.2	1.8	1.6	1.6		1.3	1.4
May 11	2.7	5.3	5.4	9.0	3.2	7.1		3.0	1.3
May 18	4.5	4.5	3.6	4.9	3.2	3.0		6.3	3.6
May 31	8.6	1.6	1.6	1.8	1.8	1.5		1.2	0.9
Jun 08	1.3	0.4	0.1			0.4		0.3	0.2
Jul 13	8.4	9.2	8.4	6.6	7.4	6.0			
Jul 19	1.6	1.7	2.3	2.6	1.9	1.6	1.3	0.1	
Jul 27	6.2	6.8	8.3	6.6	4.7	5.0	5.9	1.9	2.2
Aug 10	3.4	3.2	1.6	5.8	3.9	4.2	5.8	1.6	
Aug 17	31.5	30.7	35.1	35.4	21.6	14.0	7.5	4.3	1.7
Aug 24	6.2	10.1	20.4	18.5	12.0	10.0	8.7	1.2	1.1
Aug 31	2.0	2.1	2.3	2.3	2.4	1.6	2.9	1.0	1.9
Sep 08	13.5	15.1	3.9	3.9	4.8	4.1	3.0	4.2	1.7
Sep 21	1.6	0.9	8.2	4.0	7.2	2.9	0.8	2.2	0.2
Sep 28	1.5	6.4	4.5	2.2	2.0	5.2	5.3	4.3	1.1
Oct 05	3.1	1.9	3.2	7.6	1.3	1.8	1.9	0.3	3.5
Oct 17	0.8	0.4	0.3	1.0	1.1	1.0	0.5	0.3	0.3
Oct 26	1.7	1.0	1.4	3.4	2.4	4.5	2.2	1.4	2.0
Nov 02	2.2	2.2	1.9	2.8	2.6	6.9	2.9	2.2	1.9
Nov 09	1.4	1.3	3.2	1.7	2.1	1.4	1.3	3.4	1.3

Table A20. Dark bottle uptake in Ghost Lake (mg C m^{-3} hr^{-1})

Date	0 m	0.5 m	1.0 m	1.5 m	2.0 m	3.0 m	6.0 m	10.0 m
1976								
Sep 29	1.0	1.0	1.4		1.4	1.0		2.4
Dec 20	1.5	1.7	1.7	1.4	1.0	2.6		2.1
1977								
Jan 26	1.1	1.0	0.4	0.9	1.0	1.8		1.3
Feb 22	1.1	0.6	0.5	1.1	0.8	1.1		0.5
Mar 30	1.1	1.2	1.1	1.1	1.0	0.9		0.6
Apr 29	1.6	1.7	2.1	2.0		1.4		1.5
May 26	1.0	1.6	2.4	2.5	0.7	1.2		0.8
Jul 25	3.3	2.2	0.8	1.3	3.4	3.8		1.0
Aug 29	1.2	0.8	1.1	0.7	1.3	1.1	1.0	1.1
Oct 04	0.6	0.9	0.8	0.4	1.0	1.0	1.9	0.9
Nov 08	0.6	1.0	1.0	0.4	1.1	1.1	0.5	1.4

Table A21. Dark bottle uptake in Island Lake (mg C m^{-3} hr^{-1})

Date	0 m	0.5 m	1.0 m	1.5 m	2.0 m	3.0 m	6.0 m	10.0 m
1976								
Sep 29	0.5	0.5	0.5		1.2	0.8		0.4
Nov 01	4.3	4.3	6.6		8.6	4.5		10.3
1977								
Feb 15	1.4	2.1		1.2	1.4	1.6		1.2
Apr 29	2.4	2.3	2.5	2.9	2.9	2.2		1.9
May 24	2.7	4.3	2.4	1.8	1.0	1.6		0.4
Jul 21	3.6	3.4	5.0	5.4	4.5	4.8		4.6
Aug 29	1.2	2.2	1.2	1.1	0.8	1.3	0.8	1.9
Sep 27	1.0	0.5	0.9	0.7	1.4	1.9	1.5	1.0

Table A22. Zooplankton of Baptiste, Ghost, and Island Lakes.

Baptiste Lake

Crustacea:

Cladocera:

- *Daphnia galeata mendotae* Birge
- *Chydorus sphaericus* (O.F. Muller)
- *Bosmina longirostris* (O.F. Muller)
- *Daphnia retrocurva* Forbes
- *Daphnia pulicaria* Forbes
- *Ceriodaphnia lacustris* Birge
- *C. affinis* Lilljeborg
- *Leptodora kindtii* (Focke)

Copepoda:

- *Diacyclops bicuspidatus thomasi* (S.A. Forbes)
- *Diaptomus oregonensis* Lilljeborg
- *Acanthccyclops vernalis* Fischer
- *Mesocyclops edax* (S.A. Forbes)
- *Macrocyclops albidus* (Jurine)

Rotifera:

- *Keratella cochlearis*
- *Kellicottia longispina*
- *Polyarthra dolichoptera*
- *Pompholyx sulcata*
- *Keratella testudo*
- *Keratella quadrata*
- *Asplanchna priodonta*
- *Asplanchna girodi*
- *Filinia longiseta*
- *F. terminalis*
- *Conochilus unicornis*
- *C. hippocrepis*
- *C. natans*
- *Synchaeta pectinata*
- *Synchaeta oblonga*
- *Synchaeta sp.*
- *Trichocerca multicrinis*
- *Collotheca* sp.
- *Polyarthra minor*

Ghost Lake

Crustacea:

Cladocera:

Daphnia galeata mendotae Birge
Daphnia pulicaria Forbes (or schoedleri sars)
Ceridaphnia lacustris Birge
Daphnia retrocurva Forbes
Bosmina longirostris (O.F. Muller)
Chydorus sphaericus (O.F. Muller)
Alona rectangula Sars
Macrothrix laticornis (Jurine)

Copepoda:

Diaptomus oregonensis Lilljeborg
Acanthocyclops vernalis Fischer
Acanthodiaptomus denticornis (Wierzejski)
Eucyclops speratus (Koch)
Harpacticoid sp.

Rotifera:

Keratella cochlearis
Asplanchna priodonta
Polyarthra remata-minor group
Polyarthra euryptera
Conochilus natans
Pompholyx sulcata
Keratella quadrata
Kellicottia longispina
Filinia longiseta
Synchaeta sp.
Keratella testudo
Collotheca sp.
Trichocerca similis
Brachionus angularis
Notholca sp.

Island Lake

Crustacea:

Cladocera:

Daphnia galeata mendotae Birge
Daphnia pulicaria Forbes
Bosmina longirostris (O.F. Muller)
Ceriodaphnia lacustris Birge
Chydorus sphaericus (O.F. Muller)
Diaphanosoma leuchtenbergianum Fischer

Copepoda:

Diacyclops bicuspidatus thomasi S.A. Forbes
Diaptomus oregonensis Lilljeborg
Mescocyclops edax (S.A. Forbes)
Acanthocyclops vernalis Fischer
Acanthodiaptomus denticornis (Wierzejski)
Harpacticoid sp.
Tropocyclops prasinus

Rotifers:

Keratella cochlearis
Filinia longiseta
Keratella quadrata
Conochilus unicornis
Kellicottia longispina
Polyarthra remata-*minor* group
Polyarthra euryptera
Keratella testudo
Trichocerca multicrinis
Synchaeta pectinata
Pompoholyx sulcata
Asplanchna priodonta
Trichocerca similis
Trichocerca cylindrica
Collotheca sp.
Euchlanis dilatata
Synchaeta tremula
Hexarthra mira
Notholca labis
Brachionus angularis

Table A23. Relative Dominance of Zooplankton Species in Baptiste South.

	APR 29	MAY 5	MAY 17	MAY 19	MAY 26	JUN 1	JUN 8	JUN 15	JUN 22	JUL 7	JUL 14	JUL 20	JUL 28
Keratella quadrata	++	++	++	++	++	++	++	+		R			
K. cochlearis	+	+	++	++	++	+++	+++	++	++	R	R		R
Kellicottia longispina	+	++	++	+	++	++	++	+++	+++	++	++	++	+
Asplanchna priodonta		++	++	++	+	+	+	+	+	+	+	+	
Filinia longiseta		+	+	+	+	++	+	+	++	+	+		
Conochilus unicornis		+							+	+		+	+
C. natans			+++	+									
Synchaeta			+++	+++	+								
Polyarthra dolichoptera				++	++	+++	++	+					
Pompholyx sulcata									R		R		
Trichocerca multicrinis										R			
Brachionus angularis													
Keratella testudo	++	++	++	++	++	+	+						
Synchaeta sp.													
Collotheca													
Diacyclops bicuspidatus thomasi	+++	+++	+++	+++	+++	+++	+++	+++	+++	+++	++	++	+
Bosmina longirostris	+	+	+	+	+	+	++	+++	++	+			
Daphnia galeata mendotae	+	+	+	+	+	+	+	+	++	++	+	+	+
D. pulex	R												
D. pulicaria						R							
D. retrocurva													
Chydorus sphaericus					+	+	+	+	+	+++	+++	+++	+++
Diaptomus oregonensis							R	+	+	+	+	++	++
Acanthocyclops vernalis							+	+	+	+	+	++	++
Leptodora kindti							R	+	+				R
Diaphanosoma leuctenbergianum										+	+	+	++
Mesocyclops edax													R
Ceriodaphnia lacustris													
C. affinis													
Chaoborus flavicans	+	+	+				+			+	R	+	

R - RARE + - PRESENT ++ - COMMON +++ ABUNDANT

Table A23. Relative Dominance of Zooplankton Species in Baptiste South.

	AUG 2	AUG 11	AUG 24	AUG 30	SEP 15	SEP 21	OCT 5	OCT 12	OCT 20	OCT 27	NOV 3	NOV 10	DEC 15
Keratella quadrata										+	+	+	+
K. cochlearis		+	++	+	+	++	+++	+++	+++	+++	+++	+++	+++
Kellicottia longispina	+	+	++	+	R	+				+			R
Asplanchna priodonta				R	+	R	R		R	R	R	+	++
Filinia longiseta			R	++	+	R			R				+
Conochilus unicornis	+	++	+	+	+								R
C. natans													
Synchaeta										++	++	+	+
Polyarthra dolichoptera		R		+			R	+	+	+	R	R	+
Pompholyx sulcata										R			
Trichocerca multicrinis			R	+							R		
Brachionus angularis			+	+	+	+	+	+	++	+	+	+	R
Keratella testudo													
Synchaeta sp.													
Collotheca													
Diacyclops bicuspidatus thomasi				R	+	++	+	++	++	++	++	+++	++
Bosmina longirostris					R	R	+	+	+	++	+	+	++
Daphnia galeata mendotae	++	++	++	++	++	++	+++	+++	+++	+++	+++	+++	++
D. pulex													
D. pulicaria				R		R	+	+	+	+	+	+	R
D. retrocurva													
Chydorus sphaericus	+++	++	R		+	+	+	+++	+++	++	++	+++	+++
Diaptomus oregonensis	++	++	++	++	++	++	+	+	R	+	+	+	+
Acanthocyclops vernalis	++	++	+	+	++	++	+	+	+	+			
Leptodora kindti		+											
Diaphanosoma leuctenbergianum	++	+	+	+			R						
Mesocyclops edax	R	R	R		+	++	+				R		
Ceriodaphnia lacustris							R	R			R		
C. affinis													
Chaoborus flavicans	+	+	+	+	+	+		+					

R - RARE + - PRESENT ++ - COMMON +++ ABUNDANT

Table A23. Relative Dominance of Zooplankton Species in Baptiste South.

	JAN 5	JAN 19	FEB 2	FB 16	MAR 9	MAR 23	APR 28	MAY 11	MAY 18	MAY 25	MAY 31	JUL 6	JUL 13
Keratella quadrata	+	R	R	R	R	R	R	+	++	++	+++	+	R
K. cochlearis	++	++	+	R	R	+	+	R	+	+	++	R	+
Kellicottia longispina	+	+	+	+	+	+	+	++	+++	+++	+++	++	++
Asplanchna priodonta	++	+	+	+	+	+	+	++	++	++	+		
Filinia longiseta	+	+	++	++	+	+	+	+	+	++	++	+	+
Conochilus unicornis			+	+	+	+	++	+++	+	R	R		+
C. natans			R	+	+	+	+	R					
Synchaeta	R						+	+++	++				
Polyarthra dolichoptera	+	+	+	+	R		+	+	++	+++	+++	+	+
Pompholyx sulcata													+
Trichocerca multicrinis													
Brachionus angularis	+++	++	++	+	+	+	+		R				
Keratella testudo		+	++	++	+	+	+	++	++	+	+		
Synchaeta sp.									R				
Collotheca													
Diacyclops bicuspidatus thomasi	+++	+++	+++	+++	+++	+++	+++	+++	+++	+++	+++	++	++
Bosmina longirostris	++	++	++	+	+	+	+	+	R		R	R	
Daphnia galeata mendotae	++	++	++	+	+	+	R		R	R	R	++	+
D. pulex					R								
D. pulicaria	R	+		R						R		R	+
D. retrocurva													+
Chydorus sphaericus		R										+	R
Diaptomus oregonensis	R										R	++	++
Acanthocyclops vernalis												+	+
Leptodora kindti												+	+
Diaphanosoma leuctenbergianum												+	++
Mesocyclops edax										R	R	+	+
Ceriodaphnia lacustris												R	
C. affinis											R		
Chaoborus flavicans													

R - RARE + - PRESENT ++ - COMMON +++ ABUNDANT

Table A23. Relative Dominance of Zooplankton Species in Baptiste South.

	JUL 27	AUG 3	AUG 10	AUG 17	AUG 24	AUG 31	SEP 8	SEP 13	SEP 21	SEP 28	OCT 5
Keratella quadrata	R	R	R		+	+	R	R			
K. cochlearis	+	+	R	+	++	++	+	+	+	+	+
Kellicottia longispina	++	++	+	+	+	+	+	+	+	R	R
Asplanchna priodonta					R	R	+	+	+	R	R
Filinia longiseta					R	R	R	R			
Conochilus unicornis	++	++			+	R					
C. natans											
Synchaeta											R
Polyarthra dolichoptera	+	R	R	R	+	+	+	+	R	+	+
Pompholyx sulcata	+	R	+	+	+	+	+	+	++	R	+
Trichocerca multicrinis					R	R	R	R	R		
Brachionus angularis											
Keratella testudo											
Synchaeta sp.											
Collotheca											
Diacyclops bicuspidatus thomasi	+						R	+	+	+	++
Bosmina longirostris										R	R
Daphnia galeata, mendotae	++	++	++	++	+	+	+	+	+	+	++
D. pulex											
D. pulicaria	+	++	++	+	++	+	+	+	+	R	R
D. retrocurva	++	++	+	++	++	+	+	R	R		
Chydorus sphaericus	+	+	+	R	R	R	R	R	R	+	+
Diaptomus oregonensis	++	++	++	++	++	++	+	+	+	+	+
Acanthocyclops vernalis	+	++	++	+	R		R	+	+	R	R
Leptodora kindti		R									
Diaphanosoma leuctenbergianum	+	++	++	R	R	R					
Mesocyclops edax	+	+	R	R	+	+	+	R	R	R	
Ceriodaphnia lacustris	R		+		+	+	+	+	+	+	++
C. affinis											
Chaoborus flavicans	+	+	+	+	+	+	+	+	+	+	+

R - RARE + - PRESENT ++ - COMMON +++ ABUNDANT

Table A23. Relative Dominance of Zooplankton Species in Baptiste South.

	OCT 11	OCT 11	OCT 11	OCT 11	OCT 17	OCT 26	NOV 2
Keratella quadrata		R	R		+	+	R
K. cochlearis	++	++	++	++	+++	+++	+++
Kellicottia longispina	+	+	+	+	+	+	
Asplanchna priodonta	+	R	+	+	+	++	++
Filinia longiseta		R	R		R		
Conochilus unicornis							
C. natans							
Synchaeta	+	+	+	+	+	+	+
Polyarthra dolichoptera	++	++	++	++	+++	+++	+++
Pompholyx sulcata	++	++	++	++	+++	+	+
Trichocerca multicrinis					R		
Brachionus angularis							
Keratella testudo							
Synchaeta sp.							
Collotheca							
Diacyclops bicuspidatus thomasi	++	++	++	++	++	++	++
Bosmina longirostris	+	R	R	+	+	+	+
Daphnia galeata mendotae	++	++	++	++	++	++	++
D. pulex							
D. pulicaria	+	+	+	+	+	R	R
D. retrocurva							
Chydorus sphaericus	++	++	+	+	+++	+++	+++
Diaptomus oregonensis	+	+	+	+	+	+	+
Acanthocyclops vernalis	+	+	+	+	+	+	+
Leptodora kindti							
Diaphanosoma leuctenbergianum							
Mesocyclops edax			P				
Ceriodaphnia lacustris	+	+	+	+	+	P	
C. affinis							
Chaoborus flavicans	+		+				

R - RARE + - PRESENT ++ - COMMON +++ ABUNDANT

Table A24. Relative Dominance of Zooplankton Species in Baptiste North.

	MAY 31	JUN 22	JUL 6	JUL 13	JUL 19	JUL 27	AUG 3	AUG 10	AUG 17	AUG 24	AUG 31	SEP 8	SEP 13
Keratella quadrata	++			R		R	R		R	+	+	+	
K. cochlearis	+++	+++	++	++	++	++	++	+	++	++	++	++	+
Kellicottia longispina	+++	+++	++	++	++	++	++	++	++	++	++	++	+
Asplanchna priodonta	+	R								R	+	+	
Filinia longiseta	+	R				R	R				R		
Conochilus unicornis		+++			R	+	+	++	R	+	++		
C. natans													
Synchaeta	+												
Polyarthra dolichoptera	++	+		R						R	+	+	R
Pompholyx sulcata	+	++	+	++	++	++	++	++	++	+++	+++	++	R
Trichocerca multicrinis								R			R	R	R
Brachionus angularis													
Keratella testudo	R												
Synchaeta sp.													
Collotheca					+		R						
Diacyclops bicuspidatus thomasi	++	+	+	+	+					R	R	R	R
Bosmina longirostris		R									R		
Daphnia galeata mendotae		+++	++	++	++	++	++	++	++	++	++	++	+
D. pulex													
D. pulicaria					R	+	R	R	R	R	R		R
D. retrocurva			R	+	R	R	R	+	+	+	+	R	
Chydorus sphaericus		+		+	R	R	+	+	+	++	++	++	+++
Diaptomus oregonensis	R	+	++	++	+	+	++	++	++	++	++	+	R
Acanthocyclops vernalis	+	++	+	+	+	+	++	+	+	R	+	+	+
Leptodora kindti		R	R				R			R			
Diaphanosoma leuctenbergianum		R	++	+	+	+	+	+	+	R	R	R	
Mesocyclops edax		R		R	+	R	R	+	+	+	+	+	+
Ceriodaphnia lacustris		R		R	R						R	+	R
C. affinis													
Chaoborus flavicans				+	+	+		+	+				

R - RARE + - PRESENT ++ - COMMON +++ ABUNDANT

Table A24. Relative Dominance of Zooplankton Species in Baptiste North.

	SEP 21	SEP 28	OCT 5	OCT 12	OCT 17	OCT 26	NOV 2	NOV 9
Keratella quadrata							R	
K. cochlearis	+	+	+	+	+	+	+	+
Kellicottia longispina	+	+	R	R	R	R	R	
Asplanchna priodonta						R	+	+
Filinia longiseta								
Conochilus unicornis	R				R			
C. natans								
Synchaeta				+	+	+	++	+
Polyarthra dolichoptera	R	R	+	++	+	+	++	+
Pompholyx sulcata	R	+	+	+	+	+	+	+
Trichocerca multicrinis	R	R	R	R				
Brachionus angularis								
Keratella testudo			R				+	+
Synchaeta sp.			R	+	R	R		
Collotheca								
Diacyclops bicuspidatus thomasi	+	+	+	+	++	++	++	++
Bosmina longirostris	R		R	+	+	+	+	+
Daphnia galeata mendotae	++	++	++	++	++	++	++	+
D. pulex								
D. pulicaria	R							
D. retrocurva	R							
Chydorus sphaericus	+++	+++	+++	+++	+++	+++	+++	+++
Diaptomus oregonensis	+	+	+	+	+	+	+	+
Acanthocyclops vernalis	+	+	+	+	+	+	R	R
Leptodora kindti								
Diaphanosoma leuctenbergianum	R							
Mesocyclops edax	+	+	+	+	+	R		
Ceriodaphnia lacustris	R		R	+	+			
C. affinis								
Chaoborus flavicans	R							

R - RARE + - PRESENT ++ - COMMON +++ ABUNDANT

Table A24. Relative Dominance of Zooplankton Species in Baptiste North.

	APR 29	MAY 5	MAY 17	MAY 19	MAY 26	JUN 1	JUN 8	JUN 15	JUN 22	JUN 28	JUL 14	JUL 20	AUG 3
Keratella quadrata	+++	+	+++		++	+			R				
K. cochlearis	+++	+	+++		+++	+++		++	+	+	+	R	R
Kellicottia longispina	+	+			++	+++		++	++	++	+	+	+
Asplanchna priodonta	++	++			R	R		+	+				
Filinia longiseta	+	+			++	++		++	+				
Conochilus unicornis		+						R		R	++	+++	++
C. natans	+++				R								
Synchaeta	++	+++											
Polyarthra dolichoptera	+	++			+	+++		R			R		
Pompholyx sulcata	R	++				R					R	+	+
Trichocerca multicrinis										R	R	R	
Brachionus angularis													
Keratella testudo	++	++	++		++	++							
Synchaeta sp.	+	++											
Collotheca								R	R	+			
Diacyclops bicuspidatus thomasi	+++	+++	+++		+++	+++		+++	++	+	+	R	
Bosmina longirostris	++	+	++		+	+		+					
Daphnia galeata mendotae					+	+		++	++	++	+++	+++	+++
D. pulex													
D. pulicaria						R							R
D. retrocurva											R	R	
Chydorus sphaericus					R	R		+	+	+	+++	+++	+++
Diaptomus oregonensis					R	R		+	+	+	+	+	R
Acanthocyclops vernalis		R								R	++	++	++
Leptodora kindti								+	+	+			
Diaphanosoma leuctenbergianum										R	+	+	+
Mesocyclops edax		R			R	+				R	R	+	+
Ceriodaphnia lacustris													
C. affinis													
Chaoborus flavicans	+											+	

R - RARE + - PRESENT ++ - COMMON +++ ABUNDANT

Table A24. Relative Dominance of Zooplankton Species in Baptiste North.

	JUL 6	AUG 11	AUG 24	AUG 30	SEP 8	SEP 15	SEP 21	SEP 27	OCT 5	OCT 20	OCT 27	NOV 30	DEC 1
Keratella quadrata	R									R	R	R	R
K. cochlearis		R	++	R	R	+	+	++	++	+++	+++	++	++
Kellicottia longispina	++	+	R	R									R
Asplanchna priodonta			R	R	+						R	R	+
Filinia longiseta			R		R						R	R	R
Conochilus unicornis	++	++							R	R		R	
C. natans													
Synchaeta										+	++	+	++
Polyarthra dolichoptera	R		R			R		R	R	+	+	R	
Pompholyx sulcata	+	+	++	+	+	R	+	R		+	+	R	R
Trichocerca multicrinis	+		+	R	R								
Brachionus angularis													
Keratella testudo													R
Synchaeta sp.	R										R	R	
Collotheca													
Diacyclops bicuspidatus thomasi	+			+	+	R	R	+	+				
Bosmina longirostris													
Daphnia galeata mendotae	+++	+++	++	++	++	++	++	++	++				
D. pulex													
D. pulicaria							R						
D. retrocurva													
Chydorus sphaericus	+	+++	+	+	+	+	++	+++	++				
Diaptomus oregonensis	+	+	++	++	++	++	+	+	R				
Acanthocyclops vernalis	+	++	+	+	++	++	++	+	+				
Leptodora kindti													
Diaphanosoma leuctenbergianum		+	R	R	R	R	+						
Mesocyclops edax	R	+	R		R	+	+	+	R				
Ceriodaphnia lacustris													
C. affinis													
Chaoborus flavicans													

R - RARE + - PRESENT ++ - COMMON +++ ABUNDANT

Table A24. Relative Dominance of Zooplankton Species in Baptiste North.

	DEC 8	DEC 15	DEC 20	JAN 5	JAN 19	FEB 2	FEB 16	MAR 9	MAR 23	APR 28	MAY 11	MAY 18	MAY 25
Keratella quadrata	R	R	R	R	R	+	R	R	R	+	+	+	++
K. cochlearis	+	+	+	+	+	+	R	+	+	+	+	+++	+++
Kellicottia longispina	+	+	+	+	+	+	+	+	+	++	++	+++	+++
Asplanchna priodonta	++	+	R	+	+	+	+		+	+	+	+	+
Filinia longiseta	++	++	++	++	++	+	+	+	+	+	+	+	++
Conochilus unicornis				R		+	++	+	++	+++	+	R	
C. natans						R	R	R	R		R		
Synchaeta	+	R	+	R					R	++	+++	+	R
Polyarthra dolichoptera	R	R		R	R	R	+			+++	+++	+++	++
Pompholyx sulcata	+	+	R	+	+		R	R	R	+++	+	R	R
Trichocerca multicrinis													
Brachionus angularis													
Keratella testudo	+	+	+	+	+	+	R	+	+	++	++	++	++
Synchaeta sp.		R		R						++	+		
Collotheca												R	
Diacyclops bicuspidatus thomasi	+	+	+	+	+	++	+++	+++	+++	+++	+++	++	+
Bosmina longirostris	++	++	++	++	+	+	+	+	+	R		R	
Daphnia galeata mendotae	+	+	+	+	+	R							
D. pulex													
D. pulicaria	+	R	R	R									
D. retrocurva													
Chydorus sphaericus	++	+	+	R	R								
Diaptomus oregonensis		+	R	R								R	
Acanthocyclops vernalis	R					R				+	R	R	+
Leptodora kindti													
Diaphanosoma leuctenbergianum													
Mesocyclops edax											R	R	R
Ceriodaphnia lacustris													
C. affinis													
Chaoborus flavicans													

R - RARE + - PRESENT ++ - COMMON +++ ABUNDANT

Table A25. Orthophosphate (SRP) loadings (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1})

Stream	1976	1977	1978	x coeff.
A	15.59 (0.02)	99.81 (0.12)	153.93 (0.18)	0.11
B	4.05 (0.07)	107.25 (1.74)	24.15 (0.39)	0.73
C	25.09 (0.09)	57.95 (0.22)	41.82 (0.16)	0.16
D	19.77 (0.03)	23.89 (0.04)	58.52 (0.09)	0.05
E	236.09 (0.03)	112.93 (0.02)	646.78 (0.09)	0.05
F	84.33 (0.03)	90.27 (0.03)	254.47 (0.08)	0.05
G	1.01 -	0.96 -	1.10 -	-
H	16.60 (0.25)	2.79 (0.04)	2.62 (0.04)	0.11
I	26.19 (0.07)	57.97 (0.16)	25.45 (0.07)	0.10
J	5.94 (0.05)	50.43 (0.45)	8.51 (0.08)	0.19
K	19.56 (0.08)	21.77 (0.09)	3.52 (0.01)	0.06
L	109.31 (0.02)	169.89 (0.03)	682.66 (0.10)	0.05
M	4.19 (0.07)	16.27 (0.26)	12.97 (0.21)	0.18
N	94.32 (0.22)	49.04 (0.11)	107.84 (0.25)	0.19
O	102.02 -	315.77 -	459.44 -	-
Total A-M	567.72	812.18	1916.50	

Table A26. Organic nitrogen loadings (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1})

Stream	1976	1977	1978	x coeff.
A	159.86 (0.19)	1217.06 (1.43)	1272.61 (1.50)	1.04
B	13.64 (0.22)	255.38 (4.13)	105.04 (1.70)	2.02
C	129.29 (0.48)	435.78 (1.62)	570.85 (2.13)	1.41
D	286.9 (0.43)	822.31 (1.24)	1177.66 (1.78)	1.15
E	3803.67 (0.52)	5425.92 (0.73)	22458.66 (3.04)	1.43
F	2138.04 (0.65)	6976.36 (2.13)	11971.86 (3.66)	2.15
G	15.20 -	20.52 -	35.38 -	-
H	73.59 (1.10)	34.33 (0.51)	74.92 (1.12)	0.91
I	257.04 (0.72)	390.31 (1.09)	435.02 (1.22)	1.01
J	53.97 (0.48)	382.25 (3.43)	147.53 (1.32)	1.74
K	184.31 (0.75)	224.20 (0.91)	138.05 (0.56)	0.74
L	1345.06 (0.20)	6507.78 (0.98)	20798.97 (3.14)	1.44
M	17.30 (0.28)	68.54 (1.10)	118.63 (1.90)	1.09
N	239.26 (0.56)	694.96 (1.62)	863.02 (2.02)	1.40
O	4204.12 -	22256.62 -	38208.71 -	-
Total A-M	8477.88	22760.74	59305.18	

Table A27. Nitrate-N + Nitrite-N loadings (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1})

Stream	1976	1977	1978	x coeff.
A	6.87 (0.01)	807.79 (0.95)	378.78 (0.45)	0.47
B	0.85 (0.01)	75.14 (1.22)	14.68 (0.24)	0.49
C	13.46 (0.05)	130.73 (0.49)	46.38 (0.17)	0.24
D	43.44 (0.07)	86.25 (0.13)	60.42 (0.09)	0.10
E	461.95 (0.06)	365.76 (0.05)	1154.78 (0.16)	0.09
F	195.91 (0.06)	141.33 (0.04)	252.64 (0.08)	0.06
G	0.64 -	0.59 -	1.07 -	-
H	17.51 (0.26)	6.85 (0.10)	1.23 (0.02)	0.13
I	12.33 (0.03)	89.37 (0.25)	24.52 (0.07)	0.12
J	7.24 (0.06)	61.15 (0.55)	2.53 (0.02)	0.21
K	8.72 (0.04)	4.51 (0.02)	0.85 (0.003)	0.02
L	49.29 (0.01)	61.20 (0.01)	344.68 (0.05)	0.02
M	0.50 (0.01)	12.45 (0.20)	33.73 (0.54)	0.25
N	4.03 (0.01)	67.14 (0.16)	11.80 (0.03)	0.07
O	3223.91	1772.93	1266.89	-
Total A-M	818.71	1843.12	2316.29	

Table A28. Ammonia-N loadings (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1})

Stream	1976	1977	1978	x coeff.
A	19.07 (0.02)	52.50 (0.06)	133.87 (0.16)	0.08
B	5.25 (0.08)	36.90 (0.60)	3.33 (0.05)	0.24
C	22.30 (0.08)	42.15 (0.16)	14.55 (0.05)	0.10
D	41.74 (0.06)	64.04 (0.10)	40.49 (0.06)	0.07
E	831.15 (0.11)	596.97 (0.08)	950.94 (0.13)	0.11
F	305.52 (0.09)	618.42 (0.19)	351.32 (0.11)	0.13
G	62.52 -	72.55 -	74.94 -	-
H	13.79 (0.21)	2.87 (0.04)	1.61 (0.02)	0.09
I	42.26 (0.12)	76.29 (0.21)	40.59 (0.11)	0.15
J	8.22 (0.07)	94.10 (0.84)	5.60 (0.05)	0.32
K	28.41 (0.11)	8.07 (0.03)	2.53 (0.01)	0.05
L	274.54 (0.04)	498.90 (0.08)	858.95 (0.13)	0.08
M	3.90 (0.06)	26.81 (0.43)	3.53 (0.06)	0.18
N	44.92 (0.10)	56.47 (0.13)	61.50 (0.14)	0.12
O	602.01 -	3216.54 -	2000.35 -	-
Total A-M	1659.67	2190.57	2482.25	

Table A29. Orthophosphate (SRP) loadings (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1})

DRA	1976	1977	1978	x coeff.
1	0.94 (0.05)	17.39 (0.93)	5.42 (0.29)	0.42
2	4.46 (0.08)	54.59 (0.98)	15.60 (0.28)	0.45
3	0.51 (0.03)	0.68 (0.04)	1.53 (0.09)	0.05
4	0.26 (0.03)	0.26 (0.03)	0.68 (0.08)	0.05
5	7.06 (0.07)	16.14 (0.16)	7.06 (0.07)	0.10
6	2.50 (0.25)	0.40 (0.04)	0.40 (0.04)	0.11
7	0.78 (0.07)	2.89 (0.26)	2.33 (0.21)	0.18
8	0.95 (0.02)	5.69 (0.12)	8.53 (0.18)	0.11
9	6.43 (0.03)	8.57 (0.04)	19.29 (0.09)	0.05
10	0.71 (0.03)	0.71 (0.03)	1.19 (0.05)	0.05
11	0.12 (0.03)	0.12 (0.03)	0.20 (0.05)	0.05
12	48.34 (0.07)	179.56 (0.26)	145.03 (0.21)	0.18
13	21.31 (0.03)	7.10 (0.01)	56.82 (0.08)	0.04
Total	94.37	294.10	264.08	

Table A30. Organic nitrogen loadings (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1})

DRA	1976	1977	1978	x coeff.
1	3.39 (0.21)	51.99 (2.78)	29.92 (1.60)	1.53
2	19.50 (0.35)	160.42 (2.88)	106.94 (1.92)	1.72
3	7.31 (0.43)	21.08 (1.24)	30.26 (1.78)	1.15
4	5.53 (0.65)	18.11 (2.13)	31.11 (3.66)	2.15
5	72.65 (0.72)	109.98 (1.09)	123.10 (1.22)	1.01
6	11.00 (1.10)	5.10 (0.51)	11.20 (1.12)	0.91
7	3.11 (0.28)	12.21 (1.10)	21.09 (1.90)	1.09
8	9.01 (0.19)	67.78 (1.43)	71.10 (1.50)	1.04
9	92.15 (0.43)	265.73 (1.24)	381.45 (1.78)	1.15
10	15.47 (0.65)	50.69 (2.13)	87.11 (3.66)	2.15
11	2.54 (0.65)	8.31 (2.13)	14.27 (3.66)	2.15
12	193.37 (0.28)	759.66 (1.10)	1312.14 (1.90)	1.09
13	319.64 (0.45)	454.59 (0.64)	1882.30 (2.65)	1.25
Total	755.21	1985.65	4101.99	

Table A31. Nitrate-N + Nitrite-N loadings (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1})

DRA	1976	1977	1978	x coeff.
1	0.19 (0.01)	20.38 (1.09)	6.55 (0.35)	0.48
2	1.67 (0.03)	47.90 (0.86)	11.79 (0.21)	0.37
3	1.19 (0.07)	2.21 (0.13)	1.53 (0.09)	0.10
4	0.51 (0.06)	0.34 (0.04)	0.68 (0.08)	0.06
5	3.03 (0.03)	25.23 (0.25)	7.06 (0.07)	0.12
6	2.60 (0.26)	1.00 (0.10)	0.20 (0.02)	0.13
7	0.11 (0.01)	2.22 (0.20)	5.99 (0.54)	0.25
8	0.47 (0.01)	45.03 (0.95)	21.33 (0.45)	0.47
9	15.00 (0.07)	27.86 (0.13)	19.29 (0.09)	0.10
10	1.43 (0.06)	0.95 (0.04)	1.90 (0.08)	0.06
11	0.23 (0.06)	0.16 (0.04)	0.31 (0.08)	0.06
12	41.44 (0.06)	138.12 (0.20)	372.92 (0.54)	0.25
13	35.52 (0.05)	28.41 (0.04)	99.44 (0.14)	0.08
Total	103.39	339.81	548.90	

Table A32. Ammonia-N loadings (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1})

DRA	1976	1977	1978	x coeff.
1	0.94 (0.05)	6.17 (0.33)	2.06 (0.11)	0.16
2	4.46 (0.08)	21.17 (0.38)	2.79 (0.05)	0.17
3	1.02 (0.06)	1.70 (0.10)	1.19 (0.07)	0.07
4	0.77 (0.09)	1.62 (0.19)	1.11 (0.13)	0.13
5	7.06 (0.12)	21.19 (0.21)	15.14 (0.15)	0.15
6	2.10 (0.21)	0.40 (0.04)	0.90 (0.09)	0.09
7	0.67 (0.06)	4.77 (0.43)	2.00 (0.18)	0.18
8	0.95 (0.02)	2.84 (0.06)	3.79 (0.08)	0.08
9	12.86 (0.06)	21.43 (0.10)	15.00 (0.07)	0.07
10	2.14 (0.09)	4.52 (0.19)	3.09 (0.13)	0.13
11	0.35 (0.09)	0.74 (0.19)	0.51 (0.13)	0.13
12	41.44 (0.06)	296.96 (0.43)	124.31 (0.18)	0.18
13	71.03 (0.10)	49.72 (0.07)	63.93 (0.09)	0.09
Total	145.79	433.23	235.82	

Table A33. Zinc loadings (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1})

Stream	1976	1977	1978	x coeff.
A			4.02 (0.005)	0.005
B			0.24 (0.003)	0.003
C			1.46 (0.005)	0.005
D			2.28 (0.003)	0.003
E			110.14 (0.01)	0.01
F			76.75 (0.01)	0.01
G			0.95 -	-
H			0.16 (0.002)	0.002
I			0.76 (0.002)	0.002
J			0.15 (0.001)	0.001
K			0.10 (0.0003)	0.0003
L			47.68 (0.005)	0.005
M			0.45 (0.006)	0.006
N			1.99 (0.003)	0.003
O			120.60 -	-
Total A-M			245.14	

Table A34. Chloride loadings (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1})

Stream	1976	1977	1978	x coeff.
A	479.52 (0.54)	1203.63 (1.35)	2481.87 (2.78)	1.56
B	63.00 (0.75)	146.30 (1.74)	122.31 (1.46)	1.32
C	397.23 (1.40)	480.78 (1.69)	653.31 (2.30)	1.80
D	477.70 (0.66)	844.39 (1.16)	1252.91 (1.72)	1.18
E	11683.56 (1.30)	5892.37 (0.66)	25992.75 (2.89)	1.62
F	8525.03 (1.56)	4903.01 (0.90)	10647.60 (1.95)	1.47
G	1979.72 -	1537.62 -	1665.83 -	-
H	367.43 (4.26)	39.29 (0.46)	67.86 (0.79)	1.84
I	831.59 (2.33)	619.61 (1.73)	455.00 (1.27)	1.78
J	227.30 (2.04)	194.84 (1.75)	133.33 (1.20)	1.66
K	807.86 (2.62)	287.34 (0.93)	122.08 (0.40)	1.32
L	2533.38 (0.29)	9791.37 (1.11)	23655.80 (2.68)	1.36
M	69.27 (0.98)	102.97 (1.46)	159.97 (2.27)	1.57
N	1356.27 (1.93)	847.48 (1.20)	1766.54 (2.51)	1.88
O	10274.08 -	32953.88 -	54206.92 -	-
Total A-M	28441.59	26043.52	67410.62	

Table A35. Calcium loadings (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1})

Stream	1976	1977	1978	x coeff.
A	4739.70 (5.31)	11759.90 (13.17)	17931.02 (20.10)	12.86
B	745.50 (8.89)	1540.73 (18.36)	1662.44 (19.81)	15.69
C	4309.06 (15.19)	7210.98 (25.42)	8574.82 (30.22)	23.61
D	19141.96 (26.26)	17006.56 (23.33)	31714.58 (43.51)	31.03
E	200955.55 (22.37)	144007.51 (16.03)	482133.51 (53.68)	30.69
F	67996.82 (12.46)	67594.89 (12.39)	100566.88 (18.43)	14.43
G	1751.70 -	1745.89 -	2105.72 -	-
H	1282.00 (14.86)	275.02 (3.19)	1040.13 (12.05)	10.03
I	6720.85 (18.80)	8309.30 (23.25)	12950.72 (36.24)	26.10
J	1967.68 (17.65)	2859.01 (25.64)	3340.62 (29.96)	24.42
K	8789.45 (28.45)	2379.39 (7.70)	3558.04 (11.52)	15.89
L	45526.18 (5.15)	126161.38 (14.27)	402262.77 (45.51)	21.64
M	508.06 (7.21)	1067.55 (15.14)	1653.00 (23.45)	15.27
N	3923.54 (5.58)	7543.80 (10.72)	6916.62 (9.83)	8.71
O	172156.58 -	449517.06 -	869543.28 -	-
Total A-M	364434.51	391918.11	1069494.2	

Table A36. Magnesium loadings (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1})

Stream	1976	1977	1978	x coeff.
A	1511.80 (1.69)	3881.44 (4.35)	5369.17 (6.01)	4.02
B	147.00 (1.75)	311.55 (3.71)	357.00 (4.26)	3.24
C	965.06 (3.40)	1656.09 (5.84)	2079.11 (7.33)	5.52
D	5603.30 (7.69)	7699.01 (10.56)	8803.89 (12.08)	10.11
E	72819.52 (8.11)	46681.94 (5.20)	153830.58 (17.13)	10.15
F	23293.30 (4.27)	25608.39 (4.69)	35247.48 (6.46)	5.14
G	670.95 -	632.01 -	674.39 -	-
H	505.55 (5.86)	107.27 (1.24)	340.18 (3.94)	3.68
I	2205.37 (6.17)	2611.99 (7.31)	3875.16 (10.84)	8.11
J	718.94 (6.45)	819.43 (7.35)	1090.75 (9.78)	7.86
K	2952.64 (9.56)	988.17 (3.23)	1379.82 (4.47)	5.75
L	14279.87 (1.62)	43931.36 (4.97)	114674.97 (12.98)	6.52
M	84.25 (1.20)	306.26 (4.34)	452.25 (6.41)	3.98
N	1316.91 (1.87)	2477.13 (3.52)	2313.86 (3.29)	2.89
O	58296.79 -	156870.00 -	283099.82 -	-
Total A-M	125757.55	135244.91	328194.75	

Table A37. Sodium loadings (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1})

Stream	1976	1977	1978	x coeff.
A	899.56 (1.01)	2327.23 (2.61)	3387.71 (3.79)	2.47
B	136.50 (1.63)	256.32 (3.06)	409.85 (4.88)	3.19
C	2031.45 (7.16)	4245.45 (14.96)	5636.34 (19.87)	14.00
D	10258.23 (14.07)	8432.39 (11.57)	10013.31 (13.74)	13.13
E	106821.96 (11.89)	57879.00 (6.44)	128817.52 (14.34)	10.89
F	14817.93 (2.72)	10650.01 (1.95)	15502.33 (2.84)	2.50
G	16591.10 -	15123.44 -	16213.98 -	-
H	960.15 (11.13)	185.60 (2.15)	707.97 (8.20)	7.16
I	1901.53 (5.32)	2285.90 (6.40)	4125.13 (11.54)	7.75
J	887.68 (7.96)	618.68 (5.55)	1460.70 (13.10)	8.87
K	1433.60 (4.64)	387.75 (1.26)	635.00 (2.06)	2.65
L	30640.06 (3.47)	109422.83 (12.38)	231985.04 (26.25)	14.03
M	115.49 (1.64)	242.28 (3.44)	423.06 (6.00)	3.69
N	1499.78 (2.13)	3818.85 (5.42)	3528.84 (5.01)	4.19
O	105367.03 -	286599.18 -	526383.28 -	-
Total A-M	287495.24	212056.88	419317.95	

Table A38. Potassium loadings (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1})

Stream	1976	1977	1978	x coeff.
A	1018.84 (1.14)	4578.92 (5.13)	5711.17 (6.39)	4.22
B	110.25 (1.31)	807.69 (9.63)	393.19 (4.69)	5.21
C	753.98 (2.66)	1821.74 (6.42)	1381.57 (4.86)	4.65
D	1237.69 (1.70)	1892.10 (2.60)	3141.58 (4.31)	2.87
E	26329.43 (2.93)	11294.39 (1.26)	31912.75 (3.55)	2.58
F	7446.58 (1.36)	8384.57 (1.54)	10324.80 (1.89)	1.60
G	141.32 -	126.02 -	136.67 -	-
H	600.18 (6.95)	111.37 (1.29)	82.72 (0.96)	3.07
I	1540.59 (4.31)	1548.16 (4.33)	1356.45 (3.80)	4.15
J	359.32 (3.22)	1182.55 (10.61)	373.90 (3.35)	5.73
K	1689.83 (5.47)	828.76 (2.68)	315.03 (1.02)	3.06
L	7939.98 (0.90)	9372.10 (1.06)	26437.19 (2.99)	1.65
M	119.49 (1.69)	501.52 (7.11)	571.57 (8.11)	5.64
N	1354.98 (1.93)	1782.89 (2.53)	3013.12 (4.28)	2.91
O	22433.85 -	52817.06 -	79554.72 -	-
Total A-M	49287.38	42449.89	82138.59	-

Table A39. Sulphate loadings (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1})

Stream	1976	1977	1978	x coeff.
A	4377.36 (4.90)	14733.77 (16.50)	16615.55 (18.60)	13.33
B	441.00 (5.26)	2815.51 (33.56)	1513.30 (18.04)	18.95
C	6106.52 (21.52)	10391.20 (36.63)	9301.64 (32.79)	30.31
D	9523.01 (13.06)	15737.09 (21.59)	13502.14 (18.52)	17.72
E	76269.45 (8.49)	57181.07 (6.37)	197549.48 (22.00)	12.29
F	28404.61 (5.21)	41614.34 (7.63)	70880.56 (12.99)	8.61
G	14518.32 -	12421.13 -	14306.22	-
H	1996.03 (23.13)	200.17 (2.32)	755.49 (8.75)	11.40
I	7603.56 (21.27)	6627.10 (18.54)	9953.58 (27.85)	22.55
J	2107.06 (18.90)	2820.20 (25.29)	3762.17 (33.74)	25.98
K	5154.24 (16.69)	1580.21 (5.12)	2257.95 (7.31)	9.71
L	16330.52 (1.84)	59447.57 (6.73)	154728.14 (17.51)	8.69
M	358.10 (5.08)	939.21 (13.32)	800.78 (11.36)	9.92
N	4736.45 (6.73)	5822.37 (8.28)	5392.37 (7.66)	7.56
O	104029.88 -	228675.25 -	349221.66 -	-
Total A-M	173189.78	226508.57	495927.00	

Table A40. Silica loadings (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1})

Stream	1976	1977	1978	x coeff.
A		1665.43 (1.86)	3596.63 (4.03)	2.95
B		286.69 (3.42)	315.16 (3.76)	3.59
C		1195.92 (4.22)	1788.76 (6.31)	5.27
D		4790.84 (6.57)	6752.54 (9.26)	7.92
E	Analysis	34673.33 (3.86)	147894.48 (16.47)	10.16
F		14338.28 (2.63)	32966.44 (6.04)	4.34
G	not	595.85 -	656.31 -	-
H		110.40 (1.28)	357.21 (4.14)	2.71
I	Available	1559.12 (4.36)	2366.08 (6.62)	5.49
J		435.00 (3.90)	646.89 (5.80)	4.85
K	in 1976	517.56 (1.68)	715.18 (2.32)	2.00
L		27157.00 (3.07)	106051.56 (12.00)	7.54
M		215.37 (3.05)	425.21 (6.03)	4.54
N		1934.85 (2.74)	2176.14 (3.09)	2.92
O		22366.02 -	108786.84 -	-
Total A-M		87540.79	304532.45	

Table A41. Bicarbonate loadings (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1})

Stream	1976	1977	1978	x coeff.
A	20690.02 (23.16)	40581.11 (45.44)	63181.05 (70.74)	46.45
B	3034.50 (36.17)	4787.01 (57.06)	5697.25 (67.91)	53.71
C	18524.62 (65.30)	27388.16 (96.54)	34341.57 (121.05)	94.30
D	97532.62 (133.83)	97729.91 (134.08)	137696.71 (188.91)	152.27
E	1232235.44 (137.20)	690492.23 (76.88)	2377713.26 (264.74)	159.61
F	320962.20 (58.83)	269734.95 (49.44)	458032.19 (83.95)	64.07
G	31149.60 -	30189.24 -	33459.90 -	-
H	8861.99 (102.69)	1459.33 (16.91)	5123.76 (59.37)	59.66
I	32620.84 (91.27)	28138.76 (78.73)	49519.83 (138.56)	102.85
J	10629.66 (95.42)	10648.78 (95.50)	12615.60 (113.14)	101.36
K	41302.97 (133.71)	11905.28 (38.54)	14533.61 (47.05)	73.10
L	289753.02 (32.78)	633581.44 (71.68)	2042963.23 (231.13)	111.86
M	2557.41 (36.28)	4219.90 (59.86)	6857.01 (97.26)	64.47
N	21862.78 (31.06)	34419.19 (48.89)	33644.46 (47.82)	42.59
O	934159.38 -	2439901.90 -	4444750.60 -	-
Total A-M	2109854.70	2850856.00	5241734.59	

Table A42. Total organic carbon loadings (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1})

Stream	1976	1977	1978	x coeff.
A	4733.64 (5.30)	13475.47 (15.09)	15693.24 (17.57)	12.65
B	556.50 (6.63)	2291.28 (27.31)	1276.20 (15.21)	16.38
C	3327.44 (11.73)	6722.09 (23.70)	8943.79 (31.53)	22.32
D	10851.12 (14.89)	13374.89 (18.35)	19323.56 (26.51)	19.92
E	158359.88 (17.63)	91700.48 (10.21)	406648.60 (45.28)	24.37
F	92313.10 (16.92)	109381.68 (20.04)	213163.72 (39.07)	25.34
G	1120.62 -	664.13 -	711.38 -	-
H	2708.76 (31.39)	510.96 (5.92)	1368.94 (15.86)	17.72
I	7281.40 (20.37)	7169.41 (20.06)	10339.53 (28.93)	23.12
J	1869.50 (16.77)	3185.79 (28.57)	2843.67 (25.50)	23.61
K	6002.00 (19.43)	2873.97 (9.30)	3062.63 (9.91)	12.88
L	51711.87 (5.85)	91224.06 (10.32)	334951.93 (37.89)	18.02
M	397.13 (5.63)	1154.01 (16.37)	1324.30 (18.78)	13.59
N	7872.04 (11.18)	10541.63 (14.98)	11916.79 (16.93)	14.36
O	180380.1 -	297739.89 -	610299.71 -	-
Total A-M	341232.96	343728.22	1019651.40	

Table A43. Total carbon loadings (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1})

Stream	1976	1977	1978	x coeff.
A	7057.61 (7.90)	18986.09 (21.26)	29739.45 (33.30)	20.82
B	955.50 (11.39)	2602.87 (31.02)	2458.01 (29.30)	23.90
C	5924.19 (20.88)	9782.76 (34.48)	16684.88 (58.82)	38.06
D	22213.84 (30.48)	31516.20 (43.24)	48803.73 (66.96)	46.89
E	331495.57 (36.91)	197406.46 (21.98)	857437.75 (95.47)	51.45
F	121216.35 (22.22)	150860.88 (27.15)	307522.79 (56.37)	35.41
G	7012.36 -	6594.13 -	7525.78	-
H	3878.78 (44.95)	781.10 (9.05)	2552.14 (29.57)	27.86
I	11844.06 (33.14)	11931.88 (33.39)	21173.69 (59.24)	41.92
J	3289.45 (29.50)	4238.34 (38.01)	5652.49 (50.69)	39.40
K	11708.30 (37.90)	4315.81 (13.97)	6450.69 (20.88)	24.25
L	89868.44 (10.17)	232266.31 (26.28)	765328.64 (86.59)	41.01
M	755.03 (10.71)	1700.66 (24.12)	2719.10 (38.57)	24.47
N	11088.13 (15.76)	17459.04 (24.81)	18918.81 (26.87)	22.48
O	277744.42 -	778834.33 -	1588354.63 -	-
Total A-M	617219.58	672983.49	2074048.9	

Table A44. Iron loadings (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1})

Stream	1976	1977	1978	x coeff.
A	22.75 (0.03)	168.39 (0.19)	98.07 (0.11)	0.11
B	1.04 (0.01)	10.60 (0.13)	1.95 (0.02)	0.05
C	55.98 (0.20)	61.46 (0.22)	45.79 (0.16)	0.19
D	68.29 (0.09)	157.42 (0.22)	306.78 (0.42)	0.24
E	5971.10 (0.67)	5921.59 (0.66)	19597.44 (2.18)	1.17
F	1112.76 (0.20)	2723.13 (0.50)	5496.87 (1.01)	0.57
G	25.89 -	64.93 -	102.86 -	-
H	23.31 (0.27)	3.51 (0.04)	5.61 (0.07)	0.13
I	86.42 (0.24)	58.97 (0.16)	76.60 (0.21)	0.20
J	7.57 (0.07)	81.47 (0.73)	5.46 (0.05)	0.28
K	36.98 (0.12)	12.18 (0.04)	3.05 (0.01)	0.06
L	415.77 (0.05)	1494.74 (0.17)	4911.60 (0.56)	0.26
M	5.41 (0.08)	4.84 (0.07)	8.73 (0.12)	0.09
N	218.19 (0.31)	71.25 (0.10)	39.42 (0.06)	0.16
O	1736.72 -	1255.60 -	2114.77 -	-
Total A-M	7833.27	10698.29	30660.81	

Table A45. Copper loadings (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1})

Stream	1976	1977	1978	x coeff.
A			0.81 (0.0009)	0.0009
B			0.05 (0.0006)	0.0006
C			0.17 (0.0005)	0.0005
D			0.75 (0.001)	0.001
E			17.40 (0.002)	0.002
F			8.75 (0.002)	0.002
G			-	-
H			-	-
I			0.12 (0.0003)	0.0003
J			0.04 (0.0004)	0.0004
K			0.02 (0.0006)	0.0006
L			11.35 (0.001)	0.001
M			0.10 (0.001)	0.001
N			0.38 (0.0005)	0.0005
O			32.36 -	
Total A-M			39.56 -	

Table A46. Manganese loadings (kg yr^{-1}) and export coefficients (kg ha^{-1} yr^{-1})

Stream	1976	1977	1978	x coeff.
A			12.72 (0.014)	0.014
B			0.26 (0.003)	0.003
C			13.67 (0.048)	0.048
D			34.24 (0.047)	0.047
E			2193.01 (0.244)	0.244
F			375.32 (0.069)	0.069
G			2.55 -	-
H			0.48 (0.006)	0.006
I			11.39 (0.032)	0.032
J			0.57 (0.005)	0.005
K			0.46 (0.001)	0.001
L			979.94 (0.111)	0.111
M			3.29 (0.047)	0.047
N			6.23 (0.009)	0.009
O			694.28 -	-
Total A-M			3627.90	

Table A47. Percent distribution of nutrient loading to Baptiste Lake, 1976 (kg yr^{-1})

	Total Input	South Basin	%	North Basin	%	Outlet
Total-P	1895.03	1525.39	80.5	369.64	19.5	391.52
SRP	662.09	457.85	69.2	204.24	30.8	102.02
Total-N	11751.14	9203.44	78.3	2547.70	21.7	5134.55
Organic-N	9233.09	7189.51	77.9	2043.58	22.1	4204.12
NO$_3$+NO$_2$-N	922.10	783.29	84.9	138.81	15.1	323.91
NH$_3$-N	1805.46	1402.67	77.7	402.79	22.3	1659.67

Table A48. Percent distribution of nutrient loadings to Baptiste Lake, 1977 (kg yr^{-1})

	Total Input	South Basin	%	North Basin	%	Outlet
Total-P	2792.78	1835.19	65.7	957.59	34.3	1036.88
SRP	1106.28	650.38	58.8	455.91	41.2	315.77
Total-N	30991.01	20193.92	65.2	10797.09	34.8	28346.14
Organic-N	24746.39	16225.98	65.6	8520.41	34.4	22256.62
NO$_3$+NO$_2$-N	2182.93	1819.91	83.4	363.02	16.6	1772.93
NH$_3$-N	2623.80	1649.14	62.9	974.66	37.1	3216.54

Table A49. Percent distribution of nutrient loadings to Baptiste Lake, 1978 (kg yr^{-1})

	Total Input	South Basin	%	North Basin	%	Outlet
Total-P	5124.69	3258.50	63.6	1866.19	36.4	2122.39
SRP	2180.58	1275.50	58.5	905.08	41.5	459.44
Total-N	68871.34	43243.43	62.8	25627.91	37.2	41467.47
Organic-N	63407.17	39569.94	62.4	23837.23	37.6	38208.71
NO$_3$+NO$_2$-N	2865.19	2027.08	70.7	838.11	29.3	1266.89
NH$_3$-N	2718.07	1674.68	61.6	1043.39	38.4	2000.35

Table A50. Annual flow - weighted mean concentrations (mg L^{-1}) for orthophosphate (SRP)

Land Use	Stream	1976	1977	1978	$\bar{x}$
Agric.	A	0.13	0.25	0.32	0.23
	B	0.39	1.51	0.82	0.91
	M	0.33	0.48	0.42	0.41
	N	0.49	0.18	0.42	0.36
Forested	E	0.04	0.03	0.04	0.04
	F	0.03	0.02	0.04	0.03
	K	0.14	0.36	0.07	0.19
	L	0.08	0.05	0.07	0.07

Table A51. Annual flow - weighted mean concentrations (mg L^{-1}) for nitrate + nitrite-N

Land Use	Stream	1976	1977	1978	$\bar{x}$
Agric.	A	0.06	2.06	0.78	0.97
	B	0.08	1.06	0.50	0.55
	M	0.04	0.37	1.09	0.50
	N	0.02	0.25	0.05	0.11
Forested	E	0.08	0.09	0.08	0.08
	F	0.08	0.04	0.04	0.05
	K	0.06	0.08	0.02	0.05
	L	0.04	0.02	0.04	0.03

Table A52. Annual flow - weighted mean concentrations (mg L^{-1}) for ammonia-N

Land Use	Stream	1976	1977	1978	$\bar{x}$
Agric.	A	0.16	0.13	0.27	0.19
	B	0.50	0.52	0.11	0.38
	M	0.31	0.79	0.11	0.40
	N	0.24	0.21	0.24	0.23
Forested	E	0.14	0.15	0.06	0.12
	F	0.12	0.16	0.06	0.11
	K	0.20	0.13	0.05	0.13
	L	0.20	0.15	0.11	0.15

Table A53. Annual flow - weighted mean concentrations (mg L^{-1}) for organic nitrogen

Land Use	Stream	1976	1977	1978	$\bar{x}$	Excluding 1976
Agric.	A	1.34	3.10	2.61	2.35	2.86
	B	1.30	3.60	3.55	2.82	3.58
	M	1.36	2.02	3.84	2.41	2.93
	N	1.25	2.59	3.35	2.40	2.97
Forested	E	0.66	1.35	1.52	1.18	1.14
	F	0.82	1.83	1.94	1.53	1.89
	K	1.30	3.73	2.78	2.60	3.26
	L	0.99	1.94	2.56	1.83	2.25

Table A54. Annual flow - weighted mean concentrations (mg L^{-1}) for Nitrate-N + Nitrite-N

Land Use	Stream	1976	1977	1978	$\bar{x}$
Agric.	A	0.06	2.06	0.78	0.97
	B	0.08	1.06	0.50	0.55
	M	0.04	0.37	1.09	0.50
	N	0.02	0.25	0.05	0.11
Forested	E	0.08	0.09	0.08	0.08
	F	0.08	0.04	0.04	0.05
	K	0.06	0.08	0.02	0.05
	L	0.04	0.02	0.04	0.03